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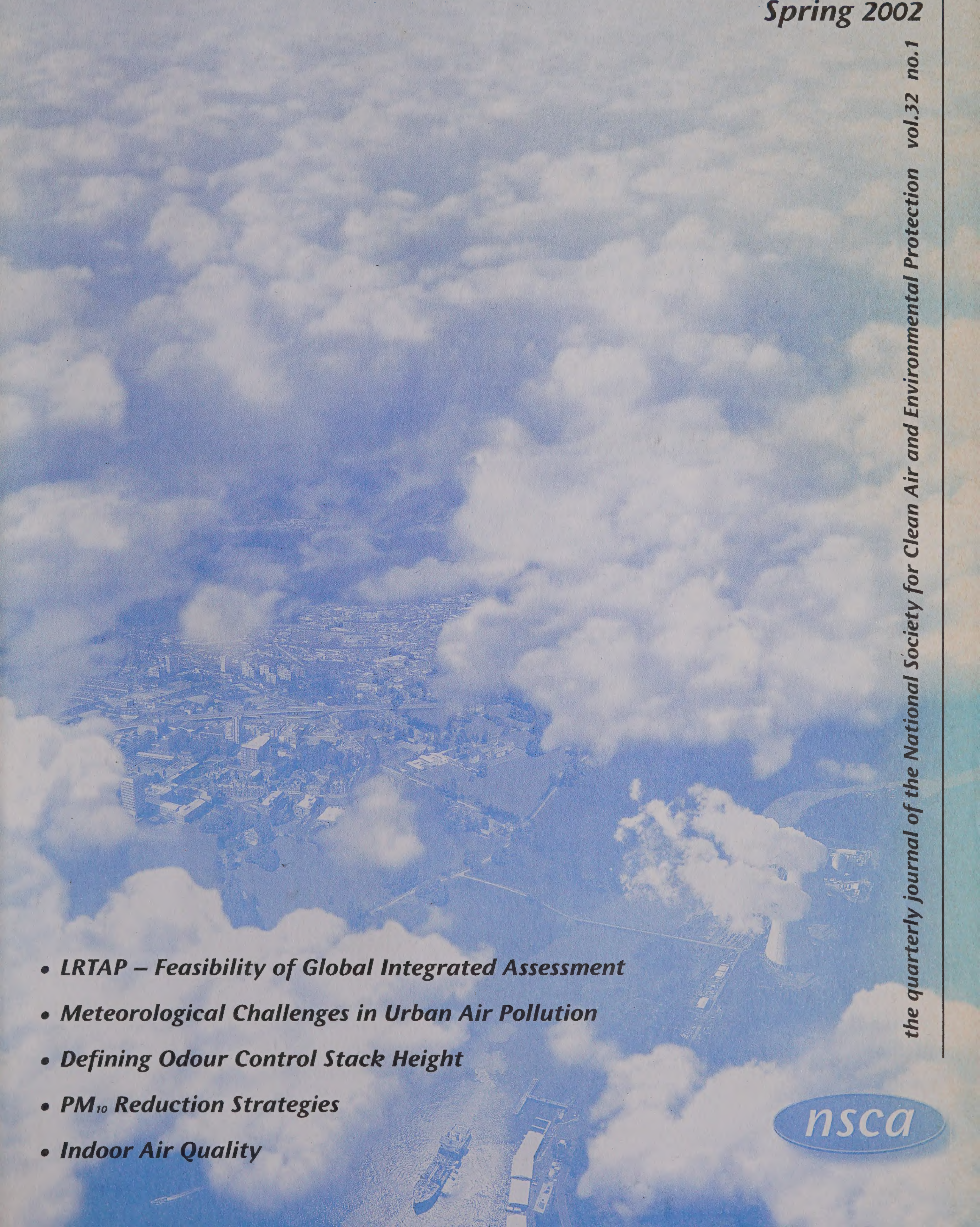

- LRTAP – Feasibility of Global Integrated Assessment
- Meteorological Challenges in Urban Air Pollution
- Defining Odour Control Stack Height
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CONTENTS

Editorial 3

Long Range Transport of Air Pollution: Report of a Feasibility Review of Global Integrated Assessment International Union of Air Pollution Prevention & Environmental Protection Associations 5

Meteorlogical Challenges in Urban Air Pollution
D.R. Middleton 14

Use of Dispersion Modelling to Define the Height of Odour Control Stacks
P.S. Childs and D.L. Hall 20

Examining the Potential for Local Authorities in Norfolk to Reduce PM_{10} Concentrations in their Areas
Dr. Tim Chatterton and Dr. Steve Dorling 25

Factors Influencing Indoor Air Quality – Quantifying the Roles of Domestic and Outdoor Sources of Air Pollutants
Steven Naylor 30

Advertiser: University of Leeds 4

The National Society for Clean Air and Environmental Protection produces information, organises conferences and training events, and campaigns on air pollution, noise and environmental protection issues. Founded in 1899, the Society's work on smoke control led to the Clean Air Acts. More recently NSCA has been influential in developing thinking on integrated pollution control, noise legislation, and air quality management.

NSCA's membership is largely made up of organisations with a direct involvement in environmental protection: industry, local authorities, universities and colleges, professional institutions, environmental consultancies and regulatory agencies. Individual membership is also available to environmental specialists within industry, local authorities, central government, technical, academic and institutional bodies.

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Editorial

A PROBLEM SHARED

In this issue, we focus on recent progress made by the International Union of Air Pollution Prevention and Environmental Protection Associations (IUAPPA), of which NSCA is a founder member and for which it provides the secretariat. It has resolved to develop ways in which the understanding of air pollution on a world wide scale can be better quantified and understood and we report here on how that important task is being addressed.

The work in Europe of the UNECE Convention on Long Range Transboundary Air Pollution has been one of the success stories of international environmental policy. However, the problems of regional ozone and particulates make it clear that the work is far from complete. Elsewhere in the world the position is far less satisfactory. In many developing countries, emissions of the main pollutants have been rapidly increasing. Long range transport means that much of their impact will be at regional and global level and institutions for collaboration within scientific, operational and political disciplines are as yet unequal to the challenges.

As one of its main priorities, IUAPPA has been undertaking a major review of scientific, technical and policy requirements at regional and global levels. In this issue's first article, we publish a summary of the reports of the preliminary stages of this project, together with the Resolution passed by last year's World Clean Air Congress on the action that now needs to be taken to address the problems.

Work on the programme is overseen by IUAPPA's Director General, Richard Mills from NSCA, and much of the work undertaken by Teresa Gonzalez of Imperial College. For further information they can be contacted at NSCA's head office.

In the remainder of the issue, we cover some of the other important topics within the air pollution field, albeit of a more domestic nature. Some of the key questions to come out of the recent round of local air quality review and assessment concerned meteorological data. In response to some of these concerns, the Met Office, along with other agencies across Europe, is investigating the use of met data in urban settings. Dr. Doug Middleton reports on the COST 715 project, and sends out a request for met observations in or near urban areas.

We also carry reports on three other issues. The first is on the use of dispersion modelling to assess chimney height for odour dispersal, a hot topic for those applying for or writing process permits under IPPC/IPC. Dr. Tim Chatterton looks at PM_{10} reduction strategies in Norfolk, uncovering an intractable problem, and Steve Naylor reports on indoor air quality in West Cumbria.

Next Issue: Local Sustainability

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IUAPPA

Long-Range Transport of Air Pollution

Report of a Feasibility Review of Global Integrated Assessment

International Union of Air Pollution Prevention & Environmental Protection Associations

This paper summarises work undertaken by the Union to explore the scope for establishing a more effective political and institutional framework, at regional and global level, to tackle the long range transport of air pollution; and to explore how far recent developments in the underlying science provide a basis for this.

BACKGROUND

The project represented a response by the International Union to a number of themes and issues which it recognised as of increasingly widespread concern, in particular:

- air pollution remains a serious source of health damage and environmental degradation and in many areas its impact is increasing. Many air pollution problems are primarily local or regional in their origin and impact, but background levels associated with the long-range transport of pollutants make a substantial contribution and need to be addressed. This requires increased co-operation at global and regional level.
- rapid decline in emissions of some of the more important transboundary pollutants within the UNECE area is being counterbalanced by the very rapid increases elsewhere, posing issues which were new in both their scale and their scope. Air pollution associated with large-scale forest burning has become a new and serious problem.
- work on stratospheric issues, and in particular their relationship to long-range transport of pollution at tropospheric level, is meanwhile changing the context in which tropospheric issues need to be addressed. Climate change has given added impetus to the global scale in air pollution policy and the need to address pollutants such as SO_2 at that level is beginning to alter the context within which regional assessments and management strategies are seen. At the same time a number of studies, principally in the United States, are beginning to reflect the need now to inter-relate the metropolitan, regional and global levels of assessment.
- meanwhile there has been rapid progress at the scientific level. 'Integrated assessment' is now established as an important tool for bringing together scientific and policy analysis in the pursuit of solutions to complex and intractable problems. It has now been applied with some success to air pollution problems at regional level within the UN/ECE Convention on Long-range Transboundary Air Pollution (CLRTAP), and there is an emerging interest in its application at other levels, including the global scale;
- as early as 1992, as part of the Rio Process, Agenda 21 called for the extension of the CLRTAP experience to other regions of the world. There is a need to review how far this is progressing.

These considerations pointed, *inter alia,* to the need for a study which could review general trends and issues and in particular review the feasibility of the wider application of integrated assessment modelling to long-range transport at the global level. However it became apparent at an early stage that if the review was to be of practical value it was important that the 'global scale' should be construed widely to cover both the possibility of a single assessment for the world as a whole, and the possibility either of separate hemispheric assessments or of a linked range of regional studies which might in relevant circumstances sum to a wider global perspective. At all stages therefore the review looked at the implications of the analysis for the separate options of a single global assessment and of a linked series of regional assessments.

It was necessary also to clarify the interpretation of the concept of 'feasibility'. It was concluded that this should also be construed as widely as practicable. Hence feasibility was taken to apply at three separate levels:

- a theoretical level, which looks at how far systems and processes not previously applied at global scale could be applied at that level;
- a practical level, which looks at the availability of data, skills and resources to deploy and calibrate any model at alternative levels globally and in other regions;
- and, finally, a political level, which seeks to assess how far outcomes would be likely to prove deliverable and effective in policy terms.

The structure of this report largely follows these three levels of feasibility:

- it first reviews the theoretical feasibility of applying the assessment process at global and regional levels, and the implications for the nature and robustness of any conclusions that might be drawn;

- it then reviews the state of technical understanding and data availability on the wide range of factors which must be played into an integrated assessment process, at both global and regional levels, including current and projected emissions, eco-system sensitivity, pollutant transport and deposition, and abatement options and costs;
- finally it discusses the key cultural and political considerations which could have a bearing on the practicality of applying the process at global level, and in other regions.

To set the scene, however, the report first reviews very briefly some of the conclusions which may be drawn from the experience, in which the UK Government has been closely involved, of the Convention on Long-range Transport in Europe. This experience represents the principal practical learning ground for the issues addressed in this report and can help to clarify some of the key considerations.

UN/ECE EXPERIENCE: THE CONVENTION ON LONG RANGE TRANSBOUNDARY AIR POLLUTION (CLRTAP)

It is to be expected that appropriate approaches to tackling the regional management of transboundary air pollution will vary significantly from one region to another. They need to reflect the cultural, economic and political circumstances of the particular region. Nevertheless, in considering the scope for progress globally and in particular regions, it is useful to have in mind as background the experience within the UNECE region to which the Resolution at the Rio Conference referred. There are several possible lessons.

First, while the problem of acid rain in Europe – on which the UNECE Convention first focused – became a significant area of international policy controversy, it was one which cut across the major political alliances and tensions in the continent. It could therefore be addressed outside the main political divisions of Europe. Indeed it was focused at its sharpest between otherwise allied countries in the Western half of the continent. Subsequently through periods of deep political controversy in Europe, the issue could be constructively pursued without jeopardising wider continental political positions.

Indeed the LRTAP Convention offered certain wider opportunities in political terms in that the countries involved, straddling the main political conflict lines in Europe, could utilise it as a relatively safe framework for continued contacts, even at periods of very high tension on other more major policies. This was made easier, however, by the fact that on this issue leadership did not naturally fall to any of the major protagonists in the then Cold War, but fell largely to Scandinavian countries, who were seen as strongly internationalist, politically less partisan, and as having a strong tradition in international bridge-building. This suggests that on this issue smaller, relatively non-aligned countries can – and perhaps must – contribute the essential element of acceptable international leadership.

The scope for progress was further enhanced by the extent to which the issue could initially be seen as an international scientific challenge rather than – primarily – as an area of conflicting political interests. Subsequently political opportunities were widened by important developments in science and practice within Europe and North America which were crucial to progress. While CLRTAP itself stimulated a wide range of scientific activity, it was also able itself to benefit from subsequent developments which helped provide a more secure base for the development of the Convention than could have been anticipated at the beginning. In sum, the early commitment of the initiative to the neutral ground of scientific assessment meant that a substantial, strong and enduring basis for the programme could be progressively assembled outside of the mainframe of policy controversy.

UNECE experience suggests however that a particularly crucial requirement is the emergence of an agreed single base model, in this case the RAINS model. This was critical to the medium and long term success of the Convention in that it provided the objective mechanism for narrowing political disagreement. Were there a large number of different models, markedly competing in their assumptions, it is exceedingly unlikely that there would have been a sufficient degree of technical consensus to allow policy progress. In the event, most other models became little more than helpful critiques at the margins of elements of the core RAINS models. In effect, the contribution of the RAINS model was to narrow options to a feasible range for negotiation, and it is arguable that it is this that was the critical factor in progress. It was arguably even more important than the emergence, at an earlier stage, of the 'critical loads' approach which had been crucial in allowing a bridge to be developed between environmental processes and policy instruments.

It seems unlikely, however, that scientific development could have been transformed into political progress without the contribution of supportive economic change. It was crucial to success that many of the contributions sought from individual Member States came to be recognised as in line with their own policy aspirations on other fronts. In particular, LRTAP almost certainly could not have been so successful were it not for changes to energy policy in European countries which automatically made the environmental gains possible.

These various considerations provide an indication of the range of factors which may be relevant to the potential success of future attempts to apply integrated assessment elsewhere and in different circumstances. They cannot be ranked in any strict sense, but include, in particular:

- a high degree of consensus on the intellectual, scientific and technical underpinning for policy debate, which in particular narrows the range of available options;
- a political climate which ensures that the issue is seen as outside any established pattern of political conflict, and one where there is leadership acceptable to all political interests;
- and most of all, compatibility of objectives and proposed measures with wider philosophical trends and economic processes, which appear to validate and give wider acceptability to them.

Clearly any such lessons from UNECE experience cannot be translated directly to other areas and periods. Application of integrated assessment at the global level or in other regions will raise a wide range of quite different considerations. However, so long as not interpreted rigidly, these conclusions from European experience can provide a helpful background for assessing feasible development elsewhere.

THEORETICAL FEASIBILITY

Against this background the various dimensions of feasibility identified earlier can be assessed.

The first issue is how far there is a common scientific and intellectual framework which could provide the basis for enhanced multi-national efforts for addressing transboundary pollution at global and regional scale, and what any such framework might imply for how global and regional processes should develop; and how far there might in principle be considerations which might point to proceeding at global or regional level.

Integrated assessment is essentially a theoretical construct which offers a framework for drawing together in a coherent manner a range of diverse factors interacting in a complex manner. In essence therefore it should offer a mechanism capable of application to air pollution transport at any level, including the global, and to any region. However the feasibility review concluded that the nature of the model will have important implications for how far it can be successfully used at different levels, and these will need to be recognised in future applications.

The key consideration is that the essential purpose of integrated assessment is to permit economic optimisation, that is to identify the least cost option for abating emissions, taking into account the range of environmental sensitivity, i.e. critical loads and levels. However, it is in general likely that the impact of emissions will be most marked in those areas relatively nearer, rather than more distant from, the source. It can be argued that it will therefore generally be more cost-effective to abate those emissions close to the area of impact which is the subject of policy concern than those more distant. The broad implication of this must be that, as a rule of thumb, the wider the area for an integrated assessment becomes, the less direct value it is likely to have in economic optimisation terms. This might, for instance, favour the regional as against the global level of analysis.

While this proposition may have some force, a number of qualifications need to be noted. In the first place, it may be qualified by marked differences of abatement cost. Even if the source and the receptor are far distant, extreme differences in abatement costs might nevertheless justify, from the perspective of the receptor, abatement activity at such distance; and currently there are, of course, gross disparities between countries and regions in abatement costs. Equally, the relative scale of the emission between areas is also likely to be significant.

A further constraint in principle on very large-scale or global exercises might appear to be the risk of greater divergences of perception and attitude, which might in turn create greater political and negotiating difficulties. In principle this seems likely, but the evidence is far from clear: it could as well be argued that distance would avoid options being narrowed by a history of local distrust.

On the other hand a number of theoretical advantages are clearly discernible from approaching integrated assessment at a global level:

- it will deal better with boundary areas;
- it may also be a more relevant context for considering increasingly significant factors such as aircraft and shipping;
- it allows a clear link to climate change;
- and it can bring to light stresses between different environmental systems which might not otherwise be evident.

Balancing these various considerations leads to the following conclusions, in principle, about the theoretical feasibility of global integrated assessment:

- there is no theoretical case which should preclude assessments being conducted at the global level;
- indeed there are a range of factors, such as boundary conditions, aircraft and shipping, and relationship to other environmental systems which, given very high levels of very long-range transport, might suggest that the theoretical case for moving straight to integrated global assessment could become strong;
- however, the central benefits of the process are likely to be diluted, above a certain level, by increasing scale. As the scope for worthwhile economic optimisation narrows, the process seems likely to be prey to the law of diminishing returns.

This may suggest the need to proceed in a balanced and integrated way at both global and regional level. The questions then are how far, at each level, the scientific and political conditions exist for progress, and what institutions might best promote this.

PRACTICAL FEASIBILITY

Tackling long-range and transboundary transport of air pollution depends on the availability of a scientific assessment system – such as integrated assessment modelling – to predict pollution impacts and guide policy intervention. That depends in turn on scientific skills and data availability across the various components of the modelling process, in particular:

- the generation of quantified emissions data and projections;
- understanding of ecosystem sensitivity to the relevant pollutants;
- capacity to measure or model long-range transport and deposition; and
- the knowledge of abatement methods and costs.

The review assessed in some depth the availability of useable data in each of these areas, at global and regional levels. The broad conclusions are briefly summarised here.

Emissions

The overall conclusion from the technical review is that quantification of emissions is now well advanced, especially in Europe and the USA, and is making relatively rapid progress in other parts of the world. Absence of emissions data would therefore be unlikely, in the next few years, to pose a significant constraint on a global integrated assessment or on the rapid expansion of regional activities.

Europe and the USA have for some years been creating annual emission inventories of the major pollutants potentially the subject of global integrated assessment (notably SO_2, NO_x, NH_3, CO, CO_2, CH_4, HM, POPs, VOCs and, more recently, particulates). National emission inventories are not common elsewhere in the world, but the experience of the RAINS-Asia study suggests that it is not a prohibitively difficult task to generate estimates of emissions for the broad purposes of the modelling processes, even if these have subsequently to be continuously refined.

Such national data sets are also now complemented by initiatives at global level. The IPPC has created a global database of greenhouse gases, the *Global Emissions Inventory Activity* (GEIA), which involves gridded global emissions inventories of the main gases and aerosols emitted into the atmosphere. The EDGAR database produces gridded emission inventories of GHG, NH_3, and pollutants causing acidification and ozone formation.

As the technical reports of the project indicate, however, this encouraging picture needs qualification on several fronts. More systematic and comprehensive data remains desirable for most regions outside Europe and North America, and everywhere there is a need for better validated emission estimates of NH_3 and ozone precursors, NO_x and VOCs, heavy metals and POPs.

Prediction will of course continue to represent a formidable challenge, even though IPPC has long-term emission scenarios for greenhouse gases in its *Special Report on Emission Scenarios* (SRES). However, while the uncertainties here mean that the integrated assessment process might need to work on a wider range of illustrative scenarios, this need not remove its value to policy.

Ecosystem Sensitivity

Here the position is less encouraging, both in relation to acidification and eutrophication, and to effects of heavy metals and POPs. The subject has been very extensively addressed in Europe (through the UNECE critical loads approach) and in affected regions of the United States (through the US's *National Acid Precipitation and Assessment Programme* – NAPAP). However, in other parts of the world there are only disparate local and regional studies that would need to be aggregated. Globally progress in this seems slow, although a good deal of impetus was given by the RAINS-Asia programme.

The position is mitigated to some extent by the existence of a number of global assessments. UNEP-RIVM has created broad maps of ecosystem sensitivity to acidification and eutrophication, which help identify areas of the world, such as central parts of South America, Western and Southern Africa, and Siberia, as areas that may potentially be affected by acidification and eutrophication. However it must be recognised that the critical loads used for these regions are full of assumptions and very often just an extrapolation from European values. There are also a number of other useful initiatives: for example, WHO has recommended guidelines for ozone for protection both of human health and crops, and levels have been mapped for Europe and the world generally to identify potential risk areas.

Overall however it is clear that ecosystem sensitivity is a much more subtle and difficult issue than emissions, and one which is much less well researched and understood outside Europe. This is particularly significant because there are some grounds for believing that ecosystems in the Southern hemisphere may be intrinsically more sensitive to acidification than in the Northern hemisphere, for complex physical and chemical reasons. The relative absence elsewhere of substantial and detailed analysis on the European level therefore represents a formidable obstacle to confidence in the early practicability of global studies, and – probably to a lesser extent – a constraint on the development of regional systems.

Long-range Transport and Deposition

As the core of the modelling process, it is not surprising that globally the picture here is complex.

Work is advanced in Europe, with the key component being the EMEP source/receptor relationships for acidifying and eutrophying pollutants, photo-oxidants, POPs and heavy metals. Further, work is now currently in hand on the creation of similar source/receptor matrices for particulates. The estimates of deposition which these have allowed provide the basis not only for the RAINS model, but also for ASAM, CASM and HARM.

In North America broadly similar studies have taken place under the NAPAP Programme, and this has generated maps of pollutant deposition for the US. There are also agreements with neighbouring countries, notably Canada and Mexico, regarding the transboundary aspects of most pollutants, especially acidifying substances and precursors of NO_3.

The position in Asia, and even more elsewhere, is more challenging. However in Asia there are now many independent studies which could merit aggregation. Indeed, when the RAINS-ASIA project calculated source/receptor matrices and deposition of acidifying pollutants for South-East Asia it reached the encouraging conclusion that the results were broadly in line with local and regional studies.

Certain studies at global level also now make a valuable contribution. For instance, the UNEP-RIVM Project on acidification and eutrophication has estimated deposition in all parts of the globe, with the results tending to show broad agreement with emerging local and regional studies. The position for ozone is not dissimilar, as a result of studies deploying the STOCHEM model.

Notwithstanding these more helpful developments, overall the unevenness and uncertainty of studies in different regions must still raise significant doubt as to the merits and practicality of assessments at global level at this stage. There is therefore in particular a need to undertake more studies in areas such as Africa and Latin America, and to ensure that so far as practicable existing and new studies are pursued on a consistent basis.

Abatement Methods and Costs

The initial expectation of the research team was that uncertainties and inconsistencies in this area would represent the most formidable constraint on global integrated assessment. In fact, the conclusion is that this may represent the least problematical area of the four. Research in the field is well developed in Europe and North America, and is capable of relatively simple extrapolation to other areas. Indeed this has already proved broadly successful within the RAINS-Asia project.

The critical consideration here is that the main emission categories are now fairly standardised, and, increasingly, the technologies are global, with the result that extrapolation can be relatively confident. It must be recognised that there has been substantial criticism of the limitations of cost-curves and of the factors taken into account in abatement costs within the UNECE context, and these are likely to be replicated elsewhere. Nevertheless, the position is, at least by comparison with ecosystem sensitivity, relatively straightforward and manageable.

Nevertheless, care is required in applying European assumptions elsewhere, and in particular in the implications drawn from them. It must be recognised, for instance, that whilst in the developed world it is possible to improve well-being without increasing pollution by investing in cleaner, more effective technologies, the scope for this may be less in the developing world, both for practical and cultural reasons.

Overview

While generalisation in this area is inevitably hazardous, the extensive reviews and assessments undertaken for the project point in the authors' view to the following general conclusions:

- there are very substantial regional variations in available data and expertise which need to be rapidly reduced. However the position is improving, and they should not now represent a significant constraint upon a substantial expansion of regional integrated assessments or impose a serious constraint on similar work at global level;
- in particular, the position in respect of emissions inventories and abatement costs offers real encouragement for the early practicality of global integrated assessment. The regional variability of data and understanding in respect of ecosystem sensitivity and, to a lesser extent, the patchiness of transport and deposition data, will however for some time continue to present problems, particularly at the global level;
- an incremental programme of regional and sub-regional studies would suffer least from technical constraints. In some regions, notably North-East Asia, data and analysis is already at a relatively high level. While in other regions the position may not be so advanced, at least there is the confidence that the level of data informing generalisation will be broadly consistent across the study areas.

The implication of this assessment is therefore that the more appropriate course through the next few years may be to encourage the development of regional assessments (particularly in regions contiguous to Europe), in order of capability, *and so long as these are progressively conducted on a common and consistent basis.* As these emerge, global integrated assessment of direct value to policy should become realistic, at least for some pollutants, particularly as further global data is generated by the Climate Change Programme.

POLITICAL AND INSTITUTIONAL FEASIBILITY

The political and institutional feasibility of extending integrated assessment of transboundary air pollution, either on a global or regional basis, is inevitably a more judgmental matter than assessing technical capability.

While there is no accepted framework for assessing political feasibility, there are a range of considerations which might be expected to facilitate it:

- a substantial measure of consensus at the relevant level, whether global or regional, on goals and objectives, and more generally on the nature of the problem;
- a wider political context which reinforces the search for agreement, for instance through internationally acceptable leadership; and through separation from more politically divisive matters;
- perhaps most essentially, perception of real and substantial mutual benefits from the process and its expected outcomes;
- finally, common policy mechanisms – allowing ultimately for differentiated obligations among participating countries – and a history of negotiating experience on which leaders can draw.

This section looks at how far these considerations are now present.

Experience So Far

It must be recognised that experience in developing of policy instruments for tackling atmospheric issues at the global level is not yet substantial. Aside from current work on climate change, it has been largely limited to stratospheric ozone depletion and the POPs Convention. Neither of these involved the complex mutualities that are central to integrated assessment, and all the key components of the process, particularly ecosystem sensitivity and abatement options, were substantially simpler than in the case of the more conventional regional transboundary pollutants. Moving from the simpler approaches of banning or restricting certain products or their use to the optimised and differentiated obligations of integrated assessment will therefore represent a major step change.

Equally, while the climate change negotiations represent an extremely complex process, with the beginnings of differentiated obligations, and with assessment and management required in respect of a relatively wide range of substances, they do not require the complex source/receptor relationships of long-range pollution, and the approach can in principle therefore be more straightforward.

The policy processes for developing integrated assessment at a global level could not therefore rely on any substantial bedrock of past experience. It would be likely therefore to raise a wide range of negotiating challenges which would need to be tackled de novo, suggesting that progress would be unlikely to be rapid.

By contrast the application of integrated assessment at the regional level could start from a more encouraging basis of policy and negotiating experience. Not only are there the specific and largely transferable skills and experience of the UNECE region, but this is reinforced by a climate in which regions are increasingly becoming an important dimension in environmental policy. The Regional Seas Agreements and the Arctic Convention are only the most notable examples.

Emerging regional arrangements for tackling air pollution are now numerous, although there is no accepted theoretical or analytical framework which allows the drawing of firm conclusions on their prospects, and outside the UNECE there is no application so far of integrated assessment. Nevertheless progress over the last decade or so provides a relatively strong base for further, incremental development.

Outside the UNECE the review was able to identify at least six developing treaty systems now addressing transboundary pollution:

- NAFTA in North America
- EANET in North-East Asia
- ASEAN in South-East Asia
- Malé Declaration in Southern Asia and parts of Oceania
- Harare Declaration in Southern Africa
- Canuelas Declaration in Southern Latin America.

Several of these are at a very early stage, and thus do not yet provide sufficient experience for any significant conclusions to be drawn, but a number of relevant pointers have emerged from the reviews so far undertaken of their work:

- The EANET framework for monitoring transboundary pollution, centred in particular on Japan, China and Korea, and so far led by Japan, has proved in some ways the most intractable. It is clear that its task has been complicated by political disagreements and the absence of relatively independent leadership. The difficulties are probably no greater than in the early stages of LRTAP in Europe, but the implication is that the rate of progress may initially not be rapid.

  However the Secretariat for the network is now moving to the UNEP Co-ordinating Centre in Bangkok. While the development process may still be lengthy, and shows no sign at this stage of moving beyond monitoring, independent leadership can, as UNECE has shown, markedly increase the chances of more rapid progress.

- In marked contrast has been the rate of progress on this issue in the contrasting cases of NAFTA and the Malé Declaration. In both cases, progress appears to have been facilitated by a wider treaty network which adopted transboundary pollution as one of the issues it could usefully tackle. In spite of the pre-eminence of the United States within North America, the area for major political disagreement was in neither case so great as to be disabling. Further, in the case of the Malé agreement, progress has been greatly facilitated by the leadership of a relatively small and neutral party, the Maldive Islands. This parallels in some ways the contribution of Norway and Sweden within Europe.
- The Harare Agreement and the ASEAN agreement show particular promise in that they are also working through established treaty or consultative mechanisms, and they currently lack dominant political players or disabling political controversies. Most important, perhaps, in these cases there are now problems, such as haze from forest-burning, which provide a new and more urgent incentive for progress.

It will be evident therefore that there are now a number of significant initiatives, potentially paralleling the UNECE LRTAP Convention, which could begin to make a significant impact on regional transboundary pollution problems. Furthermore, it is notable that in one particular case, the Malé Convention, there is now an explicit and agreed move towards the adoption of integrated assessment as the basis for future work.

In short, the implication of the review is that there is now the opportunity for more substantial and sophisticated development of regional treaty systems for managing long-range transport of air pollution than had previously been evident.

Experience so far suggests however that for further progress in these and other areas two factors will be crucial:

- it is important that the areas for which any further systems are proposed are 'natural' regions which have some coherence, defined by the prevalence of a specific and shared air pollution system problem and also by some wider natural unity. It would certainly be a mistake to advocate carving up the globe into regional systems, rather than basing progress on the assessment of need and opportunity in relevant areas;
- however, such an incremental approach will need support. There will be an increasing and substantial need for some degree of global co-ordination which ensures the necessary degree of consistency and compatibility of data and approaches and facilitates joint co-operation. The

need for this is elaborated further later, but in essence it points to a new function for UNEP or some other relevant international body.

Common Philosophy

A necessary factor to underpin an enduring treaty system is some degree of common philosophy and perspective. How far is this now present?

Clearly this is less likely to be present at global level, and it is notable that those regional treaty systems that have so far developed have been within regions of relative cultural homogeneity. This is not surprising, because concepts of the environment and the terms in which it is valued will largely dictate how problems are defined and opportunities perceived. However circumstances are changing.

A relevant example is differences of approach in the perception and valuation of the 'environment' between the LRTAP area and, for instance, Southern Asia. In the latter area the environment has not been a significant category of policy, and indeed philosophical perception of it is markedly different from what has emerged in recent decades in Europe. It is likely that this would have represented a significant constraint on collaboration by those two regions within a common framework.

A second example is the differences in the underlying objective among systems. For instance, within the LRTAP region the objective was to manage and abate existing emissions. By contrast, within Asia the policy objective associated with integrated assessment is far more likely to be the management and regulation of new development.

While such factors may currently represent constraints on co-operation and progress at global level, circumstances are likely to change. For instance, in both Europe and Asia the focus is increasingly on the shared problem of health effects, rather than environmental impact. Equally, the global philosophy of sustainable development, to which both Europe and Asia subscribe, may increasingly provide a common framework within which both abatement and development strategies could be relevant.

For the immediate future, distinctive regional attitudes and values may point to the merits of focusing support for the development of integrated assessment mainly on relatively homogenous existing regional networks. However, the emergence of common philosophies and priorities, such as are represented by health and by sustainable development, is likely increasingly to point towards the potential for a more co-ordinated globalised approach.

Mutual Benefit

Even more important than practical experience and philosophical convergence is likely to be perception of mutual benefit. This needs to go beyond simply the recognition of a shared problem. This is at the core of the integrated assessment system, reflected in the process of economic optimisation. Potentially, therefore, the systems are likely to be most practicable when there is a reasonable balance of benefit and disbenefit to all individual participants. Inescapably, in any such process, however, some participants are likely to benefit relatively more than others. In Europe it was evident that substantial progress was only achieved when those called on to make the relatively high contributions could see them as nationally beneficial and in line with policy developments that they might in any case have wished to pursue.

Of themselves, such considerations may also weigh against a global system, given both the scale of the disparities and the complexity and range of potential relationships. This may also limit the prospects for progress in some regions – such as North-East Asia – where there are dominant pollution sources and where the political disagreements remain significant, even though it is in just these regions that in due course a policy process based on integrated assessment could potentially be most beneficial.

At regional level recent experience further suggests that progress is likely to be most rapid where there is no single Member State as a predominant emission source, and where all or the great majority of the Member States of the network are in a broadly similar position. It is on this basis that more progress might be expected in the shorter term in Malé and ASEAN regions than with EANET in North-East Asia.

The current debate about emissions in Asia is relevant here. Arguments in favour of an early attempt at global integrated assessment might well develop added urgency and force if emissions in India and China were to rise at or above forecast levels, if evidence were to firm up on the extent of inter-continental transport between Asia and North America, and if there was a perception that more significant damage from long-range transport was as a result occurring. Fortunately the likelihood of this scenario appears to be diminishing, as emissions in China at least appear to be being successfully reduced below forecast levels. However, if in the event pressure for a global integrated assessment and policy process were to emerge on this account, the considerations discussed earlier in this paper suggest that, even if the technical work was well resourced, the pre-conditions for progress at policy level might initially be difficult to secure. While the global level is significant, abatement of Indian and Chinese emissions is even more important in a national and regional context. At those levels it may also offer greater prospect of mutual advantage than as part of a wider exercise motivated by perceived impacts on the wider world.

In sum therefore it is important that if an integrated assessment and negotiating process is to be successful, all the participants must not only see a significant problem but also must see some benefit from the proposed process. The conclusions of CLRTAP, as reflected in the Gothenburg Protocol, have already demonstrated the monetary, health-related and social benefits of action at regional level, and this may suggest that, in the first instance, the focus might most appropriately be on strengthening regional action around the world rather than immediately focusing on the global level.

However it is likely also to be the case that, in those areas such as North-East Asia, where there are significant difficulties in the face of agreement, the evidence of a movement towards a wider global framework for addressing long-range pollution would offer useful reinforcement to attempts to make progress at regional level. On this basis, for example, the development of a UNEP-led global centre for long-range air pollution management to support and co-ordinate regional networks could strike the optimum balance of regional and global concerns.

CONCLUSIONS

Overall, the various strands of the feasibility review point to a number of broad conclusions on which it is suggested that international development policies in this field should be based in the next few years.

- The long-range and transboundary transport of air pollutants remains an important issue, and in major respects more important than has so far been recognised.
- Climate Change programmes and recent atmospheric research both suggest that pollutants can be transported farther and on a greater scale than had previously been generally recognised, with important implications for health and the global environment.
- The issues are now diverse and widespread. In developed countries, as local sources of pollution are abated, the imported pollutants from distant sources become a more significant component of overall pollution, and a residual and intractable problem. Elsewhere in the world pollutant emissions are rising, for example in China and India, by some 50% over the next 20 years, potentially giving rise to air pollution transportation issues of a new scale and complexity. In some areas, recent phenomena such as large-scale forest burnings have led to regional atmospheric problems of an intense and international character.

These considerations point to the need to develop a new international framework for managing and abating air pollution in a proportionate and economically optimised way. This will present a major challenge at both a scientific and political level, but a variety of developments in the last few years suggest that there are major opportunities for progress.

At the scientific and technical level:

- The application of integrated assessment modelling in the UNECE region, and subsequently in Asia, has provided the basis for a broad consensus on the planning policy needed to address the scientific, economic and political aspects of the issue in a coherent manner.
- The state of scientific understanding and of data availability on the main components of such an assessment process (emission projections, eco-system sensitivity, pollutant transport and deposition, and abatement methods and costs) suggests that there is now wide scope for the application of more effective scientific and policy processes in different regions around the world. Outside Europe and North America, in particular, there remain significant problems associated with the understanding of eco-system sensitivity and of pollutant transport and deposition, but the scope for rapid expansion of scientific and policy initiatives is substantial. An immediate move to an integrated global assessment would raise serious difficulties and uncertainties, but the scope for such an assessment is likely to progressively increase.

In response to these opportunities, a range of regional networks have begun to emerge, in addition to UNECE, and NAFTA in North America. These include the EANET, ASEAN and Malé networks in Asia, the Harare Declaration in Africa, and the follow-up to the Canuelas Declaration on the Control and Prevention of Atmospheric Pollution (in MERCOSUR countries). It is important that such developments be supported and encouraged by an effective international framework, which facilitates the sharing of information and experience, and provides a coherent basis for linking the various regional systems.

While both the global and regional dimensions of the issue point to the desirability of some more coherent international machinery to help to promote effective progress, there is a question as to whether this should be by early movement towards the single global mechanism on the basis of a single integrated modelling process, or whether the focus should rather be on encouraging the emergence and development of regional networks. In practice, both should be pursued. On the one hand, a global approach will have benefit in terms of dealing with boundary areas between systems, including international shipping and aircraft, and the identification of stresses between different environmental systems. On the other hand, the proportionately far greater scope for effective economic optimisation at the regional level and the differences in priorities and in state of preparedness between regions, point to progress at that regional level as the first priority.

Overall, therefore, the review suggests that the most timely and effective way forward would be to develop some new international capability which could at the same time:

- encourage and support the development of regional networks;
- promote activity and data assessments in those areas where such systems might not emerge;
- promote effective co-ordination between regional networks in scientific, technical and data systems;
- prepare the ground for the development of a more fully integrated global assessment on the basis of pre-existing and consistent regional initiatives.

The LRTAP Convention of the Economic Commission for Europe has been one of the most successful international environmental Conventions. There is no iron rule to suggest that similar developments in other regions (or indeed its application at a global scale) would necessarily be equally successful. However, by comparison with most other environmental policy, it would appear to offer particularly substantial scope for progress at both scientific and policy level. This suggests that the original proposal approved ten years ago at the Earth Summit was well judged, and that it would be timely now to press that proposal forward in more concrete terms.

Resolution on Worldwide Action on Long-Range Transport of Air Pollution

At its 12th World Clean Air and Environment Congress, held in Seoul, Korea, from 26-31 August 2001, the International Union of Air Pollution Prevention and Environmental Protection Associations (IUAPPA) agreed on the following Resolution:

The IUAPPA

Recognising that

Air pollution continues to impose significant damage to the environment, public health and materials, resulting in considerable human, social and economic loss to individuals and society worldwide;

Damage associated with air pollution undermines prospects for sustainable development;

Emission reductions achieved in one part of the world may be partly offset by increases elsewhere;

Air pollution, travelling globally across national borders, must therefore be addressed in an integrated way at local, regional and global level;

Action on acidification, eutrophication, and tropospheric ozone formation has important inter-linkages with other global issues, such as programmes on climate change and persistent organic pollutants;

Noting that

Science is now in place to address air pollution on a global level;

Regional co-operation within the framework of the 1979 UNECE Convention on Long-range Transboundary Air Pollution has been successful in curbing emission of air pollutants in Europe and North America;

Further intergovernmental co-operation in worldwide integrated assessment of air pollution and abatement options, which recognises the need for cost-effective solutions and international burden-sharing, could yield significant benefits to all parties;

And that the uneven distribution of resources and capabilities in the world impairs the prospects for quick progress on emission inventories, effect studies, technology applications and associated policy development;

Calls upon

All Member States to give the issue high priority and, in particular, to support the further development of international co-operation on regional scientific and political mechanisms;

Invites

The United Nations Environment Programme (UNEP), drawing on its global experience, to establish, with the assistance of other relevant bodies, an effective international co-ordinating framework to promote relevant science and technology, encourage regional collaboration on transboundary air pollution and to elaborate necessary burden-sharing mechanisms, including integrated assessment modelling;

Resolves, in the meantime, to

Promote co-operation among relevant governmental and non-governmental organisations (NGOs) in support of establishing such a body;

Develop a programme for public and political awareness-raising, aiming at facilitating worldwide policy action to reduce emission of air pollutants and their resulting negative impact on the environment and human health; and

Assist in promotion, co-ordination and capacity-building for the development of a more effective global framework to address air pollution.

For more information about IUAPPA and its work on transboundary pollution and other issues, please contact Richard Mills, Director General, International Union of Air Pollution Prevention and Environmental Protection Associations, 44 Grand Parade, Brighton BN2 9QA. Tel: 01273 878772; Email: washton@nsca.org.uk

Research

Meteorological Challenges in Urban Air Pollution

D R Middleton

Met Office

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ABSTRACT

The European Action called COST is an international programme of co-operation in science and technology. This paper considers the work of COST 715, Meteorology Applied to Urban Air Pollution Problems. It is a request for information *concerning meteorological observations in or near urban areas. Readers from local authorities, universities, or other organisations, are invited to complete the electronic questionnaire. The results will be assembled into a European data base of urban meteorological observing stations. The work of COST 715 will be of especial interest to users of dispersion models. It is leading to new recommendations on how airport meteorological data should be adapted for urban modelling, and on how urban meteorology should be observed routinely for managing air quality. Web sites are given.*

1. INTRODUCTION

Following the London Smog of December 1952, when some ≈4000 excess deaths occurred, and the ensuing investigations of the Beaver Committee, the Clean Air Act was passed in 1956. This introduced Smoke Control areas. In December 1991, long after local use of coal had been largely eliminated across London, a severe episode of nitrogen dioxide was recorded. This also exhibited excess deaths, albeit a smaller number of ≈160. When the Bill to create the Environment Agency was passing through Parliament, the House of Lords with NSCA assistance, proposed and successfully saw Part IV added to the Bill, so that local air quality management as we see it today became a statutory duty under the Environment Act 1995.

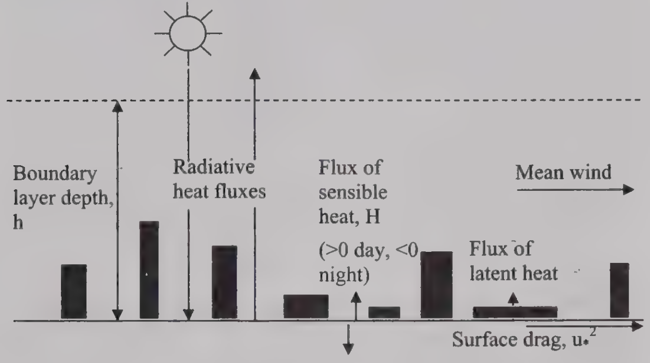
The Met Office has long been involved in advising on meteorological conditions during periods of high air pollution, and since the Chernobyl disaster, its work on dispersion of materials in the atmosphere gained renewed importance. The NAME model has come from this work and is now the UK's most advanced dispersion model for long range atmospheric pollution. It has recently been adapted for forecasting air quality in the national Air Quality Bulletins. In its role as the national Meteorological Service for the UK, the Met Office was asked to provide a representative on the COST 715 Action. This Action is particularly appropriate, in that following its earlier work on the Aeolius model for street canyons (Buckland, 1998; Buckland and Middleton, 1999; Middleton, 1999; Manning et al, 2000), the Office had been conducting field experiments in Birmingham (Ellis and Middleton, 2000a; 2000b). The measurements of urban meteorology in 1998, 1999, and 2000, were designed to improve our understanding of how urban effects might be included when modelling air quality. This was a research question of some importance, for a number of local authorities had sought advice from the Office regarding meteorological data and observations in urban areas. It had also become clear from work on the forecasting of urban air quality using the Boxurb model, that a proper diagnosis of the heat flux and atmospheric stability was needed in cities (Middleton, 1998). At the time of the Environment Act in 1995 and the launch of local air quality management, there were gaps in the scientific knowledge of urban meteorology. They needed addressing in order to answer questions of practical import. This paper summarises some key ideas on how meteorological data are used for dispersion models. It then outlines some notable achievements from the COST programme, followed by the four topics being studied by the Working Groups that make up COST 715. This paper ends with a *request for help in completing the survey* of urban meteorological observing stations.

2. URBAN MET DATA IN DISPERSION MODELS

In urban areas, the wind over increased surface roughness creates greater mechanical turbulence. Stability is influenced by the urban heat island and its effect on thermally induced turbulence, which may lead to a delay in the onset of the evening transition to night time stable conditions (Middleton, 1998). In the case of a large city these effects can lead to an almost complete absence of stable conditions, which would be present in the surrounding rural areas (Oke, 1987), and this will have an effect on air quality. A large urban heat island may generate local mesoscale flows, changing wind speeds and directions over the city.

However, the parameterisation of urban meteorology is not well understood. In order to develop urban dispersion models this must be remedied, particularly the diagnosis of stability over a town or city. Observations and physical models play a large part in developing the understanding upon which numerical models are built.

Figure 1 illustrates the factors to be considered when describing urban meteorology:



1. Mixing height *h* may not be sharply resolved, or it may be concealed by multiple inversion layers, making it poorly defined, even though it remains a useful modelling concept for limiting plume rise and vertical dispersion.
2. Buildings alter radiative properties by changes in albedo (fraction reflected) and vertical structure of spaces between buildings provides shade and radiation trapping. Buildings alter thermal properties, storing solar energy and releasing it as sensible heat late in the day, which delays the onset of evening stability and maintains vertical dispersion. Positive H denotes upward sensible heat flux (convection).
3. Heat flux to or from the ground changes with surface material: concrete, tarmac, soil etc.
4. The wind experiences obstacles; wakes, flow reversal, enhanced turbulence, and changes in the local profiles in the roughness sub-layer, as discussed by Roth (2000) and Rotach et al (2001). A measure of surface roughness length is z_0.
5. Anthropogenic energy use can be a noticeable fraction of annual solar input (≈ 10% in London) and influences the local stability of the air.

Meteorological pre-processors (Middleton, 2001) use the input data to distinguish convective situations ($H>0$; $L<0$) from stable situations ($H<0$; $L>0$), where *L* is a measure of the stability, usually assumed uniform across a city. Cities change z_0 and u_*,but also need modifications (Middleton et al., 2000) to amend *H* and *L* (which is rarely done in current models). The city may be biased towards $H \geq 0$ when compared with an airport or rural synoptic meteorological station. The stability *L* may have a limiting value in a model to prevent a model city atmosphere becoming too stable, but to what should this limit be set? If the temperature stratification is multi-layered, what should *h* be? These are the sorts of question for which current work in urban meteorology seeks answers. It requires a mixture of large scale urban experiments, small scale physical wind-tunnel modelling, and work to develop new parameterisations and to validate numerical modelling.

3. AN OVERVIEW OF COST

Co-operation in Science and Technology across Europe has a long and distinguished history, and many topics have been studied. COST is 30 years old and was reviewed by Joffre (2001): the present article draws upon his review (and personal communication). COST is an agreement between nations in Europe. It is managed by a Technical Committee; Sylvain Joffre from the Finnish Met Service chairs the COST Technical Committee for Meteorology, and serves with COST 715. This means that meteorological topics, including air pollution, are well covered within COST. The 700 series of Actions cover Meteorology in its many guises; currently eight are active. Actions run for a period and then finish, usually culminating in a final report with recommendations. The COST Action supports meetings, study contracts, visiting experts, short term scientific missions and reports. COST serves to develop ideas and promotes science through bringing experts together from different countries. By interchanging ideas on a subject and future directions for research it can have a significant impact on European science. It can lead to the writing of research proposals, often involving institutes from several countries as joint partners. COST itself does not fund research work, yet by facilitating co-operation it has led to some notable achievements. These include:

1. The European Centre for Medium Range Weather Forecasting (ECMWF) is 25 years old and grew out of COST Action 710 from 1971-73.
2. It had a role in the evolution of weather radar networks under COST 72-76; these radars are very important for modern short range forecasting as they reveal the movement of precipitation across Europe in a dramatic pictorial fashion.
3. The satellite sounder AMSU which has a role in initialising numerical weather prediction models grew from discussions in COST 712, an Action on microwave radiometry.
4. COST has constructive links with EUMETNET and EUMETSAT, stimulates activities that are basic to meteorology, and assists the development of observing, forecasting, research and training in meteorological areas.

Ultimately, developments which lead to improved numerical weather forecasts enable better air quality forecasts to be made. Dispersion models such as the Met Office NAME model rely very heavily on having the best possible predictions of the evolving weather fields. The air quality community is dependent in part upon the advancing science of meteorology. In addition to improving the basic science, there is also a role for the COST Action 715 that looks specifically at meteorological data for urban pollution. Urban air quality cannot be properly managed without due consideration of the use and suitability of meteorological data and its pre-processing within a city environment. Similarly, secondary pollutants such as particles and ozone are formed and removed during long range transport, so mesoscale meteorology becomes important for their control.

Some 20 countries across Europe have therefore signed up to the COST 715 Action on Meteorology Applied to Urban Air Pollution Problems. The Chair of the Management Committee is Professor Bernard Fisher, and the author is the second UK representative. Ranjeet Sokhi at the University of Hertfordshire is a member of the Working Group looking at episodes. The aim of COST 715 is:

> To improve the use of meteorology in understanding, modelling and managing air pollution, especially in urban areas.

It has a Management Committee (B Fisher) and Working Groups:

1. Urban wind fields *(M Rotach, Switzerland):*
   - comparing urban and rural wind measurements from European cities;
   - parameterisation of the urban wind profile using insights from wind tunnel and field experiments.
2. Heat flux and mixing height *(M Piringer, Austria):*
   - urban surface energy balance (as this affects stability diagnosis in urban dispersion);
   - urban boundary layer parameterisations;
   - mixing heights.
3. Pollution episodes *(J Kukkonen, Finland):*
   - report on episodes in European countries.
4. Meteorological input (data/pre-processors) for dispersion models *(M Schatzmann, Germany).*
   - Reviewed current practice for preparing data for dispersion models in several European national weather services;
   - Requesting help for completion of survey across Europe of urban meteorological (and air pollution) observations/monitoring.

Request for Information

The author as UK representative hereby asks all local authorities, and other institutes such as Universities, who conduct urban or near urban meteorological observations, to provide information to the COST 715 urban station survey (web site at end of this paper for electronic submission of your information please). The Met Office will provide its summary of the urban stations that it supports. When complete, this survey represents an opportunity to see how meteorological information is gathered across Europe for dispersion modelling in environmental impact analyses and local air quality management. Decisions on whether or not to declare an air quality management area can be sensitive to meteorological data. Along with emissions, meteorology plays a significant part in contributing to the overall uncertainty of urban dispersion modelling. COST 715 is seeking an improved understanding in this subject. By completing the survey, readers can contribute in a very positive way to this work. It will be valuable to know about the measurements, how the data are processed, and what they are used for, such as direct input into a dispersion model. For example, the Airviro dispersion model uses a local urban site to obtain wind and stability data. When readers have tried the electronic survey, they should contact the author with any comments or suggestions, to be reported back to the COST Management Committee.

The station survey covers measurement. As hinted at earlier, numerical weather prediction models play an increasingly important role in describing meteorological situations as they evolve in time and space. COST 715 is also looking at how such models may be adapted to take account of the urban surface as their lower boundary condition. COST 715 also examines how the outputs of weather models might be post-processed to include urban effects and render the data suitable for input to a dispersion model.

4. URBAN EXPERIMENTS

Measurements in urban areas are designed to identify ways in which local meteorology may be changed by a conurbation. It is well known that urban areas have several effects:

1. Changed thermal properties: building materials reflect/absorb radiation differently, have different thermal capacity, and the space between buildings can be shaded from the sun.
2. Changed water properties, with less evaporation and increased run-off over impermeable surfaces, but also watering in use.
3. Changed surface roughness effects causing turbulent wakes, flow reversals, vortices, and a different wind profile with height.
4. Changes in the surface characteristics over the horizontal plane, such as tarmac, concrete, parks, buildings and gardens; horizontal homogeneity breaks down over such a mixed surface.

One direct consequence is the greatly increased complexity of trying to define a good location in height and position for meteorological instrumentation in a city. Typically, a mast might need to be twice building height or more, but in some cities this can become impractical. It also means that when sensors are placed high above the surface, they 'see' a different surface according to wind fetch. Also, as the wind speed varies, the region on the surface that influence the sensor may also change, because properties like heat, momentum, or water vapour must diffuse upwards in order to reach the sensor, but at the same time these properties are advected horizontally by the wind. An important design feature of urban measurements should therefore be the parallel operation of a reference station outside the city, so that the data can be related to and compared with the more normal standard observations. There is therefore scope to apply remote sensing techniques, such as lidar (Vaughan and Forrester, 1989; Constant et al, 1989) or sodar, in order to gain information from heights and locations not attainable via masts. It is also necessary to compare the results of urban experiments with the predictions of numerical weather models, to find ways to improve the modelling of local effects. Such experiments are complex to design. They require intensive instrumentation and considerable resources. The data can take some years to analyse and report. COST 715 has been reviewing urban winds

(Rotach et al, 2001; Soriano et al, 2000) and heat fluxes (Middleton et al, 2000).

4.1 Birmingham, UK

The purpose of this study is to increase our understanding of urban meteorology, with the ultimate aim of providing an improved assessment of urban boundary layer stability, which could lead to better air quality forecasts. It is also important to have measured values for the roughness length z_0 in UK cities. This is an important variable for dispersion models. Martano (2000) had suggested a scheme to obtain surface roughness length z_0 from a single ultrasonic anemometer, whilst Grimmond et al (1998) had also discussed the derivation of aerodynamic roughness for urban areas. They cited an earlier review of 50 studies that estimated z_0 from different methods.

Measurement campaigns were conducted in Birmingham, UK, during April/May 1998, January/February 1999, and July/August 2000. The location was Dunlop Tyres Ltd., a large tyre factory. Standard meteorological observations were also taken 8.5km away from the Met Office station at Coleshill, just outside the city. Instrumented masts of 15m, 30m, and 45m were set up, secured by guy ropes. At these heights, measurements were taken of turbulence, wind and temperature. Derived from these were sensible heat flux, friction velocity, roughness length and Monin Obukhov length. Short-wave and long-wave radiation, pressure, rainfall and humidity were also measured. Since the exchange of heat between atmosphere and underlying surface is important, temperatures were also measured at 2m above the surface, on the surface (grass or concrete) and down to 1m below the surface. Results from the first two trials were described by Ellis and Middleton (2000a, 2000b). The urban temperature was ~1°C greater than the rural temperature, with some lag. Rooney (2001) has shown the roughness length was in the range 0-2m, fairly typical of values used in dispersion models for cities, with uncertainty of circa ±1m. He analysed the effects of wind direction, according to fetch and land-use type. The 2000 trial is being analysed; it had the addition of a sonic anemometer at 15m at Coleshill, so the heat flux and turbulence can be compared at the rural and city sites. It was noticeable that the urban heat flux was much nearer to zero overnight than at the rural site. These results are being used to investigate ways of improving the diagnosis of atmospheric stability; they have shown deficiencies in modelling the early morning transition.

4.2 BUBBLE, Basel, Switzerland

The Basel urban boundary layer experiment (Bubble) measures the turbulence and energy balance in the flow over the city (Rotach, personal communication). The aim is to take measurements continuously for 1 year, over 2001-2. The instrumentation includes several 30m towers with ultrasonic anemometers, and remote sensing equipment (profiler, lidar and sodar). A tethersonde or tethered balloon will also be used to lift instruments well above the surface. Model studies include urban parameterisation in mesoscale modelling, and dispersion studies using Eulerian (grid-based solving of advection and dispersion) and Lagrangian (particle random walk) models.

4.3 ESCOMPTE, Marseille, France

The region around Marseille in Southern France emits ≈ 25% of volatile organic compounds in the whole country. It has winds influenced by the land-sea breeze system with the Mediterranean, and winds coming down the valley to the city. There is a significant photochemical pollution problem. ESCOMPTE was set up to measure validation data for numerical models of pollutant transport and chemical reactions (Mestayer, personal communication). The aim was to build a data-base of emissions inventory, meteorological observations, and air pollutant concentrations. Equipment includes aircraft, a ferry, remote sensing, constant density balloons, and the surface sites The scale of this experiment is appreciated from the planning timescale:

- 1996-9 Planning & building.
- 2000 Pre-campaign: testing & preliminary data.
- 2001 June/July Main campaign.
- 2001-2003 Data processing/analysis/modelling by project team.
- 2003 Data open for general research community.

And from the resources deployed:

- some 30 surface sites for chemistry/meteorology.
- many teams, involving ≈ 200 scientists, with ≈ 100 on-site during campaign.
- Use of 7 instrumented aircraft, with 250 flight hours
- Operations cost (excludes salaries) ≈ $1.5m.

This represents a major and well planned attempt to obtain the type of comprehensive data set over a large city that modern air quality modelling requires for its development and validation.

4.4. Radio Tower Data: Hamburg, Germany; Helsinki, Finland

Radio towers meet the criterion that they can be many times average building height. They are therefore very useful platforms for mounting anemometers and thermometers. However care is needed to position the sensors as far out from the structure as possible, and to select data when the wind is approaching the sensor directly and is not in the lee of the structure.

The Hamburg radio tower is 300m tall, with measurements at 6 levels from 50m to 280m. There is an arm at each platform to hold the sensors away from the tower. Using an ultrasonic anemometer the wind components are sampled 10 times per second, whence $2x10^{14}$ samples, or 27.3 minutes, is sufficient to calculate turbulence spectra and other quantities such as turbulent length-scales. These were used in the new wind tunnel to ensure proper scaling of flow over a 1:400 (by size) urban model (Schatzmann, personal communication).

The Helsinki (Kivenlahti) radio tower has measurements up to 327m. Data have been recorded continuously from 1989 till the present. The observed wind profiles can be influenced by the presence of the mast. However the data are very valuable when analysing severe inversion conditions leading to pollution episodes in the city of Helsinki (Karppinen, personal communication). Temperature measurements enable the calculated temperatures from numerical weather prediction models (such as HIRLAM) to be validated. From 27-29 December 1995, such an episode occurred. The tower data showed the temperature inversion was reaching 18°C at its worst on the 28th. This is a much stronger temperature inversion than the severe NO_2 episode in London on 13 December 1991, when the Crawley ascent suggested the inversion in the lowest layer was of order 5°C.

The proper modelling of inversion strength and depth is very important for episodes in conurbations, when low level emissions become trapped and their vertical mixing is suppressed. There is a need for such measurements in London.

5. CONCLUSIONS

This paper presents an overview of the factors that are involved in urban meteorology. It has explained the work of COST, a co-operation between nations in Europe. COST 715 aims to develop new recommendations for the description of urban meteorology for use in urban pollution. Local air quality management throughout Europe stands to benefit from the outcomes of this work. Current researches into urban atmospheric chemistry will also benefit from better data on the urban atmospheric physics; ideally both chemistry and meteorology should be combined into focussed joint research projects. New remote sensing techniques offer scope to acquire measurements that have hitherto been unavailable. The Met Office research programme benefits from the studies in this co-operative Action.

Readers with urban stations are especially asked to make their contribution known (within 1 month of this publication please) through the COST 715 urban station survey via the web site. If difficulties arise with this, or other comments are warranted, please contact the author on doug.middleton@metoffice.com.

Web sites

Urban Station Survey:
http://www.mi.uni-hamburg.de/cost715/form.html

Preliminary Survey Results:
http://www.mi.uni-hamburg.de/cost715/inventory.html

COST 715:
http://www.dmu.dk/Atmospheric Environment/cost715.htm

Met Office home page (for Aeolius, nitrogen dioxide polynomials, Boxurb)
http://www.metoffice.com/environment

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Use of Dispersion Modelling to Define the Height of Odour Control Stacks

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ABSTRACT

Throughout the ongoing capital investment period, Southern Water is making a significant investment in the provision of odour control to satisfy planning conditions. Although much is made of engineering out odours, Southern Water has had to rely heavily on utilising abatement technologies such as wet chemical scrubbers, biofilters and carbon units and discharge treated air to atmosphere through a stack.

Historically stack locations and heights have been considered as an architectural detail, designed to minimise the visual impact and so not interfere with the planning process. As a result short stacks, of less than 5m, located alongside taller structures have become commonplace. This combination has resulted in poor dispersion and therefore the requirement for extremely low emission concentrations from the abatement system. To achieve these concentrations additional odour control has often been required, mainly the addition of carbon polishing units to the primary scrubbers.

Southern Water began an investigation to determine whether odour control costs could be controlled by appropriate design of the discharge stack. It was found that existing guidance, the UK D1 and the USEPA GEP, do not adequately address the design issues that were being faced. Consequently Southern Water has developed a new methodology to optimise odour control stack heights.

This paper was first presented at the Odour Emissions from Wastewater Treatment Processes: Monitoring Prevention and Control, October 2001, organised by Aqua Enviro.

Key words: Air Quality, Cost Benefit, Dispersion Modelling, Odour Control, Stacks, and Stack Heights

INTRODUCTION

Southern Water has a commitment to ensuring that it minimises the environmental impact as a result of its activities and that it will adopt best practicable means to minimise the likelihood of odour nuisance from its wastewater treatment works. Consideration is given to the potential for odour release at all stages of the design process. Rising mains are designed to mitigate the potential to generate septic conditions; low odour release technologies combined with engineering out areas of turbulence are used to reduce the opportunities for odour release and plants are operated to minimise the retention time of sludge. Combining these features helps to provide works with low odour release.

However, despite our best efforts we are often required to install odour abatement technologies. In such scenarios sources are covered and air is extracted to an abatement system where the air is treated and discharged to the atmosphere through a stack. Because odour control is an expensive process significant time is given to optimising the design. Particular care is given to the design of covers to ensure that containment and ventilation rates are correct, ductwork is designed to ensure they do not fill with condensate and the plant is designed to give the required performance. However, the discharge stack is an element of the treatment design process that has been largely ignored.

One potential explanation for why odour discharge stacks have not passed through the same design process as other discharge stacks is the perceived restrictions of the planning process. As a result of public concerns of wastewater treatment developments, local authorities use planning conditions to ensure that the local population will not be adversely affected. As a consequence water companies have rigorous planning conditions imposed on aspects of the development including noise, odour, landscaping and building heights. The rigour of the planning process has meant that water companies have been reluctant to challenge perceived wisdom and consequently the result has often been stacks of less than 5m in height being located alongside large structures.

PHYSICS OF DISPERSION

So why are we concerned about having low stacks next to tall buildings? The answer – a combination of cost and process reliability.

The physics of dispersion are complicated. However there is a correlation between the height of the discharge and the maximum release concentration to ensure a given ground level concentration is not exceeded. The taller the stack, the higher the concentration which can be released, and the lower the sophistication of the abatement technology which can be employed. However, as with most things, it's not that simple. There are a number of parameters that will have significant impacts including temperature of release, efflux velocity and areas of turbulence around the point of discharge.

Temperature

The release temperature is generally not a parameter that can be modified economically to provide greater dispersion. The other problem with modifying temperature is that there is a risk of generating visible plumes that have their own problems.

Efflux Velocity

Efflux velocity, or release velocity is one of the parameters that can be modified readily to achieve greater dispersion. This modification however operates within a relatively narrow band of between 10 m/s to 20 m/s. Below 10 m/s the effectiveness of dispersion rapidly drops away, and above 20 m/s there is a risk of creating whistling. Noise associated with stack discharges can create a greater problem than the impact of odour, and consequently we do not design stacks to resolve one nuisance to replace it with another.

Building Downwash

The presence of buildings near an elevated stack discharge changes the airflow patterns in the area. One aspect of these changes that is particularly important for stack dispersion is the phenomenon known as downwash. Downwash turbulence can result in the stack plume grounding earlier than would normally be the case. Because the plume has a shorter distance to disperse, the result of downwash is a higher ground level concentration.

The downwash effect can be mitigated in two main ways; either move the stack away from the downwash structures, or raise the stack above the influence of the downwash structures. Each downwash structure produces a downwash effect that radiates a distance equivalent to five times the lesser of the structure height or projected width. By mapping these zones of influence it is possible to determine a position which is outside of the downwash effect. This technique allows the zone of influence to be calculated for all wind directions. The stack height required to escape the zone of influence is 2.5 times the height of the downwash structure.

EXISTING GUIDANCE ON DISCHARGE STACK HEIGHTS

Southern Water has an ongoing program of investment optimisation. As part of this, work was undertaken to determine whether efficiencies could be realised through stack discharge design. Initially this work was targeted at new developments about to pass through the planning process but has subsequently been applied to existing sites for retrospective amendments.

The first stage of the review was to look at the existing guidance supplied by the regulatory authorities both in the UK and in the USA. In the UK guidance for determining the height of stacks is given in the document D1[1] with modifications for odour control stacks in "Odour Control and Measurement – An Update"[2]. The USEPA Good Engineering Practice (GEP) provides current guidance on stack height determination in the US[3].

To gain an understanding of how existing height determination guidance compares with Southern Water's current practice a review was required. For the review of these assessment procedures a reference site located on the North Kent coast was selected. The reference site contains a complex arrangement of tanks and buildings, 18 in total of varying heights. The emission parameters assessed were a discharge with a volumetric flow of 20,000m^3/hr and a hydrogen sulphide loading of 50 ppm that would typically have been discharged through a 5m stack.

Technical Guidance Note D1

The Technical Guidance Note D1 was produced to supplement the third edition of the 1956 Clean Air Act Memorandum on Chimney Heights[4]. The technical guidance addresses a wider variety of emissions including non-fossil fuel combustion and polluting discharges from a wide range of emissions. Odour control stacks, however, were not one of the polluting emissions given consideration within the guidance. This omission was subsequently addressed when clarification of the use of the technical guidance D1 in determining the stack height of odour control equipment was provided in "Odour Control and Measurement – An Update". This was achieved by amending the guidance to include a pollution index system. It should however be noted that problems were noted in the addendum concerned with defining stack heights for odour control using this method, with reports stating that the stack height can be either too high or too low. In addition to the recommended stack height, the technical guidance also gives guidance for the overriding minimum stack height. This is based on the heights of buildings and uncorrected stack heights. The calculation is relatively involved and has not been described in this paper, for further information please refer to the referenced guidance note.

Using the Technical Guidance Note, the recommended minimum stack height for the reference site was determined to be 32m, with an overriding minimum of 19m. These compare to a typical stack height for such a reference site of 5m.

Guideline for Determination of Good Engineering Practice (GEP) Stack Height

Guidance from the USEPA is designed to calculate *"the height necessary to ensure that emissions from the stack do not result in excessive concentrations of any pollutant in the immediate vicinity of the source as a result of atmospheric downwash, eddies or wakes which may be created by the source itself, nearby structures or nearby terrain obstacles"*. As with D1, the GEP has its origins in industrial emissions, and is not directly intended for use with odour control stacks.

The USEPA GEP does not include any provision for thermal induced buoyancy, but attempts to make an allowance for the downwash effects of buildings by recommending that the stack height be increased to outside of this zone by using the following calculation:

$Hg = H + 1.5L$

Where Hg = Stack height measured from ground level.
H = Height of nearby structures measured from ground level.
L = Lesser dimension of height or projected width of nearby structures.

GEP also accommodates an absolute minimum stack height of 65m from the base of the stack.

Using the GEP methodology for the reference site, a 20m stack is recommended, although the absolute minimum should be 65m.

PROPOSED METHODOLOGY

Typically odour control specifications would require the maximum emission concentration to be determined from a fixed set of parameters which would generally include a stack height. During our investigations it became clear that we were ignoring an area which could significantly reduce the costs of odour control. Instead of using dispersion modelling as a means to determine performance guarantees we explored the possibility of using it as a design tool.

The first part of the design process was to determine the optimum location for the odour control stack. The results of this exercise meant that the stack was required to move some 50m from its original position. The principle of determining this location was to draw a balance between the availability of services while ensuring the stack was moved from the zone of influence of the predominant downwash structures.

The second process was to investigate the relationship between stack height and the maximum permissible concentration. This work was conducted using the ISCST3 dispersion model. Separate runs were made for stack heights between 0 and 40m, at 1m intervals with and without the influence of buildings on plume dispersion. The results of this exercise are given in Figure 1.

The results presented in Figure 1 demonstrate the relationship between the stack height and the maximum permissible emission and it is clear that there is not a linear relationship between them. There is a clear difference in the rate of change of emission concentration with respect to height, with the benefit gradually tailing off. It is clear that buildings have a significant influence on dispersion, even when the location has been selected to mitigate the effect. The influence of buildings inhibits dispersion, lowering the maximum emission concentration, until the height is sufficient to discharge above the zone of downwash influence. With the reference site the height required to overcome the downwash effect is 32m, the same height which results from the D1 calculation.

The results from the reference site enabled us to propose two procedures for stack height determination:

- ***Optimum Height Determination.*** The use of modelling to determine the height of a stack to gain the maximum benefit from increasing the stack height.
- ***Technology Change Height.*** The use of modelling to determine the point where increasing the stack height further will have no additional cost benefits.

Optimum Stack Height

It was clear from Figure 1 that the rate of change altered and consequently the obvious step was to differentiate the graph produced for our reference site. The first derivative of the curve *(dy/dx)* shows the maximum permissible emission per meter increase in stack height as shown in Figure 2. The figure shows the greatest increase in maximum permissible emission per meter to occur at 27m (depicted as point A), i.e. the height at which the most benefit is being obtained from increasing the stack height.

Height for a Change of Technology

The next step was to determine the height of the stack that corresponds to enable the technology to change. The higher the percentage removal required the more sophisticated the technology required and the greater the costs of that system. For the reference site we had an inlet concentration of 50 ppm and below are details of the technology required for each range of maximum emission concentrations:

0 – 200 ppb	– Wet chemical scrubbing with a carbon polisher.
200 – 500 ppb	– Wet chemical scrubber alone.
500 – 1000 ppb	– Enhanced biofilter with a polishing system.
1000 ppb and above	– Enhanced biofilter alone.

Figure 3 depicts the points on the curve where these technology changes take place, with the main points being summarised below:

stack height range	suitable technology
0m to 16m	Wet chemical scrubber + polisher
17m to 29m	Wet chemical scrubber only
30m to 40m	Enhanced biofilter + polisher
› 40m	Enhanced biofilter only

It is clear that a typical stack height of 5m would require the most stringent levels of odour control in this reference site, however increasing the stack to 17m and then again to 29m will enable the level of technology to be reduced.

COST BENEFIT ANALYSIS

To determine whether such increases in stack height can be justified it is necessary to establish the potential savings from moving from one technology to another. To compare the whole life cost of the various odour control systems a 20 year NPV (Net Present Value) has been performed and the results displayed below. These figures are based on a system treating 20000 m³/hr at a load of 50 ppm:

Stack height range	Technology required	20 year NPV
0m – 16m	Wet chem + polisher	£ 2,400,000
17m – 28m	Wet chem only	£ 2,100,000
29m – 40m	Biofilter + polisher	£ 1,200,000
› 40m	Biofilter only	£ 880,000

By ensuring that a 17m stack is installed removal of a carbon unit from the odour control is possible, with obvious cost savings associated both in terms of capital and operational expenditure amounting to £300,000 over the NPV period. To increase the stack by 12m would cost an additional £10,000, and consequently the benefits are clear.

The benefits of increasing the stack are even greater when the stack is raised to 29m. This enables a biofilter to be installed which has significantly lower operating costs, which over the ten year period would amount to savings in excess of £1,000,000. It has to be recognised that building small diameter stacks of such heights will require additional engineering, and so will be more expensive, however the saving clearly justifies such a decision.

SUBSEQUENT APPLICATIONS

Since the initial work was conducted for the reference site the scope of application has been extended to determine whether benefits could be realised on existing installations. The first step was to perform an examination of existing odour control equipment where carbon scrubbers are currently installed, and determine whether savings could be obtained through increasing their height. These initial targets were selected with the view that removing the carbon-polishing unit would deliver operational cost savings in the current financial year. Other applications examined were in situations where the odour control system may have been under performing, and the stack height increase could address the situation without the need for significant additional investment.

Some of these have been depicted in the following case studies.

CASE STUDIES

Case Study 1. Mid-Kent

This site currently employs a three stage scrubbing unit, with a catalytic iron filter followed by a biofilter with a carbon-polishing discharging through a 6m stack to atmosphere. The system is working to the designed level and has successfully addressed the complaint issues that existed.

The site was modelled as described previously and a graph of emission concentration in relation to stack height was produced and is shown in Figure 4. The first assessment was to perform an optimum stack height analysis through the differentiation of the graph obtained. The optimal height is the point at which dy/dx is at its greatest; this is indicated in Figure 4 using the reference B, which is at a height of 38 m.

The second assessment was to determine the height required to enable a technology change, moving from a system requiring polishing to that where polishing can be removed. This level is also indicated in Figure 4 and is at a height of 11m.

Pushing for the optimum point in this example is clearly not required as an increase to 11m enables savings to be obtained from the removal of carbon.

Case Study 2. Hampshire

This site utilises a three stage scrubbing unit comprising a catalytic iron filter followed by a biofilter and carbon-polishing unit, discharging to atmosphere through a 5m stack. This site has recently been refurbished to address a significant odour problem.

Again, as with the first case study, the site was modelled as described previously and a graph of emission concentration in relation to stack height was produced and is shown in Figure 5. The first assessment was to perform an optimum stack height analysis through the differentiation of the graph obtained. The optimum stack height, where dy/dx is at its greatest, indicated as point C, with a height of 20m.

The second assessment was to determine the height required to enable a technology change, moving from a system requiring polishing to that where polishing can be removed. This level is also indicated in Figure 5 and is at a height of just 8m.

In this case study a small increase in the stack height from 5m to 8m may have the effect of increasing the emission sufficiently to enable the site to dispense with the carbon unit, and associated operational costs whilst continuing to comply with the planning condition.

CONCLUSIONS

The data presented in this paper clearly demonstrate the savings that can be realised through ensuring that the stack is appropriately positioned, and is of an adequate height.

Existing methods of defining stack heights for odour control plants, D1 and the USEPA GEP, are often not applicable for odour control stacks and as such a new method was required.

The method developed by Southern Water uses dispersion modelling as a design tool to determine locations and heights, not just emission concentrations. Through plotting the results of a dispersion modelling exercise to define the maximum permissible emissions for a range of heights, the stack designer has the scope to undertake two assessments.

- ***Optimum Height Determination (OHD).*** The use of modelling to determine the height of a stack to gain the maximum benefit from increasing the stack height.
- ***Technology Change Height (TCH).*** The use of modelling to determine the point where increasing the stack height further will have no additional cost benefits.

The examples we have examined to date have suggested that in practice it will be the TCH that is selected from this process. This method will deliver the greatest whole life cost saving. It also has the advantage that it generally ensures that a shorter stack can be installed than with OHD.

In detailing existing methods, and the development of new methods we have not touched on the problems that designing taller stacks can generate. By installing stacks over 15m, planning permission will have to be gained as it falls outside of the scope of permitted development. However our experience has been that when the reasons for such requests are made, local authorities will give serious consideration to the request.

Ensuring that odour control is designed in an integrated fashion means that environmental performance can be achieved while controlling costs. Lower technology systems are cheaper to install and operate, and in the long term are more environmentally sustainable.

Figure 1. Maximum permissible emission against stack height for the Reference site

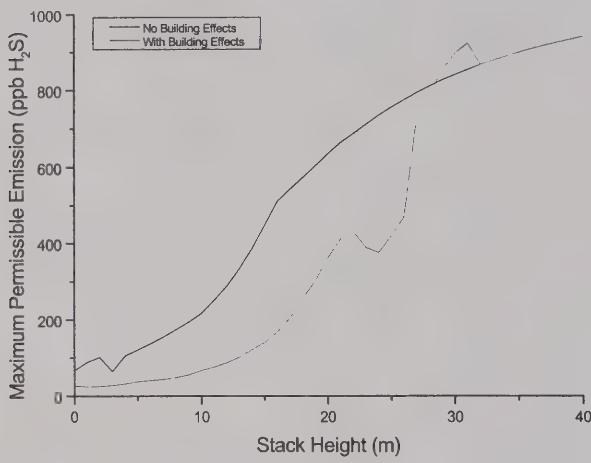


Figure 2. First derivative of the maximum permissible emission against stack height for Reference site

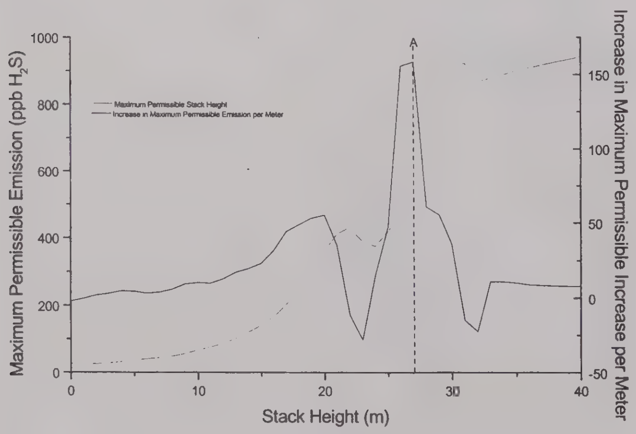


Figure 3. Height of technology change for the North Kent Coast scenario

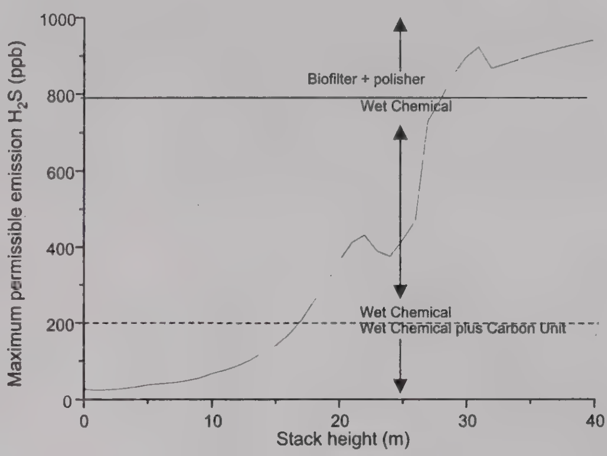


Figure 4. Maximum permissible emission against stack height, with the first derivative and technology change for the Mid Kent site

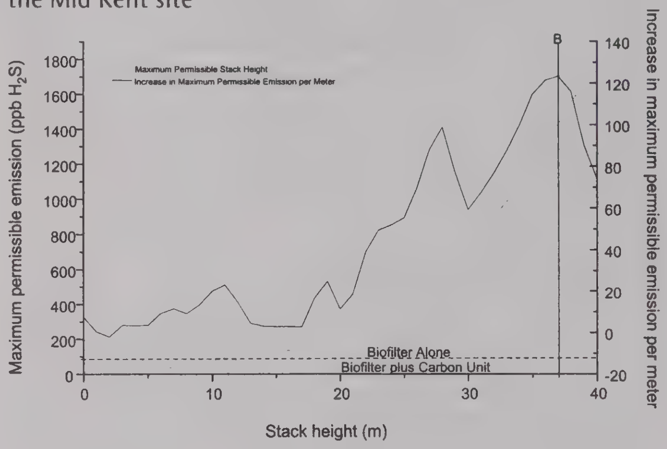
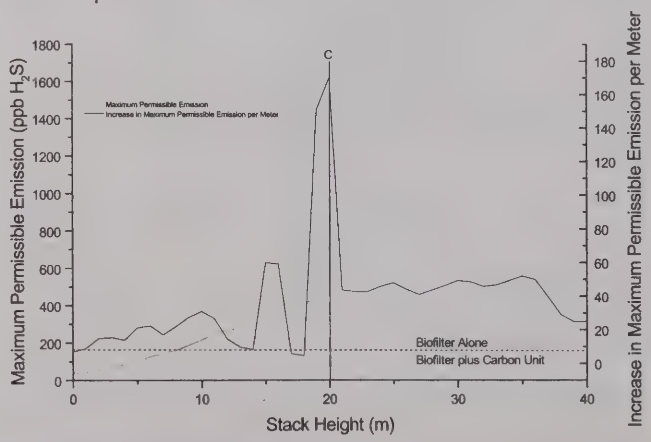


Figure 5. Maximum permissible emission against stack height, with the first derivative and technology change for the Hampshire site



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Examining the Potential for Local Authorities in Norfolk to Reduce PM_{10} Concentrations in their Areas

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ABSTRACT

This paper describes the key findings of a PhD thesis entitled "Regional and Urban Scale Modelling of Particulate Matter: Can PM_{10} be Managed at a Local Level?" (Chatterton, 2001). The PhD was jointly funded by the Natural Environment Research Council and Norwich City Council. Additional funding for the regional modelling was provided by a consortium of local councils comprising Breckland District Council, Great Yarmouth Borough Council, Kings Lynn and West Norfolk Borough Council, North Norfolk District Council, Norwich City Council, South Norfolk District Council and Norfolk County Council. The full thesis is available for downloading in PDF format at http://www.uea.ac.uk/~e083/Abstract.htm

WHY IS IT NECESSARY TO CONSIDER PM_{10} COMPOSITION?

In order to manage particle concentrations so that exceedances of air quality standards can be better understood and avoided in the future it is necessary to know two things about PM_{10}. Most importantly, the cause of exceedances needs to be known, for example the degree to which elevated concentrations are caused by primary or secondary particles. Also, typical PM_{10} composition needs to be understood so that the most effective way of avoiding exceedances can be identified as prevention of sudden elevations of certain components, or reduction of baseline levels.

QUANTIFYING PM_{10} COMPOSITION

There are two ways in which the composition of PM_{10} can be quantified: modelling and chemical analysis. Modelling of PM_{10} is a difficult task due in part to the different component species. In order to look at PM_{10} as a whole two different scale models are often used to predict the different (precursor) sources, one looking at locally generated particles and the other at the formation of regional scale secondary particles. Whilst quantified estimates of the emissions of primary particles and the precursors of secondary particles are available for most of Europe (albeit with less than ideal accuracy and spatial resolution), emission inventories of coarse particles have, at best, only been estimated very crudely.

Chemical analysis of particle composition can be time consuming and expensive. There are a great many techniques available for analysis, each with their own disadvantages. Also, no single technique can successfully analyse all the potential components and many involve the destruction of the sample preventing full analysis of all components.

THE PM_{10} PROBLEM IN NORFOLK

Under the initial air quality objective (50μg/m^3 as the 99th percentile of daily maximum 24-hr running means) Norwich had a problem. Monitoring of PM_{10} began at Norwich Centre AURN (possibly more accurately described as 'Urban Background' than 'Urban Centre') at the end of July 1997 and by mid-November that year had already registered 5 exceedances of the objective level. These were on a single day on 18 August and around the end of October/beginning of November when there were four days (30/10/97- 2/11/97) where the daily maximum running 24 hour mean exceeded the 50μg/m^3 level (see Figure 1). The maximum daily means over these five days were: 50.4, 66.9, 75.5, 55.5 and 51.1 μg/m^3 respectively. If the data is compared with the aerosol sulphate observations made at the Stoke rural sulphur network site (especially if a multiplication factor of 2.45 is applied to represent the total secondary aerosol (as measured by TEOMs) as suggested by APEG) it is clear that the two episodes are of different nature. The August episode appears to be a classic summertime episode driven by photochemical generation of secondary aerosol whilst examination of meteorological conditions during the October/November episode (see Figure 2) shows low temperatures and very low windspeeds that are typical of the meteorological conditions frequently associated with wintertime episodes. However, this latter event is notable for its timing as it occurred over the weekend before Guy Fawkes' night when both Halloween and bonfire/fireworks events are likely to have contributed significantly to particulate levels across the UK. As such it is an excellent example of why the 99th percentile was incorporated into the air quality objective for PM_{10}.

Due to a lack of monitoring outside Norwich, nothing at that time was known about parallel levels of PM_{10} across the rest of the county of Norfolk. However, following the issue of TG4(98), which indicated that all urban areas in England should move to a Stage 2 Review and Assessment, a group of Norfolk councils funded modelling work to assess the potential for controlling PM_{10} through local management measures. This work used the UK Meteorological Office's NAME model (Maryon, 1991) to consider the PM_{10} experienced across the county. This modelling was also supplemented by:

(i) PM_{10} monitoring data from Norwich Centre AURN site;

(ii) PM_{10} and $PM_{2.5}$ data collected by Norwich City Council's mobile monitoring unit;

(iii) further PM_{10} and $PM_{2.5}$ monitoring and chemical analysis data from the rural West-Norfolk Stoke Ferry site (collected as part of a six-month DTI/ETSU funded project (King et al, 1999));

(iv) urban-scale modelling using ADMS-Urban. All monitoring data was collected using TEOM instruments.

MONITORING RESULTS

Figure 3 shows the relationship between daily mean PM_{10} at Norwich Centre and Stoke Ferry between March and September 1998 (the period of data collection overlap). Readily apparent from the graph is the fact that the concentrations follow each other very closely (the correlation coefficient, *r*, between the two PM_{10} series is 0.889). Over the whole six-month period the concentrations are very similar with a mean difference of just over 2µg/m³, or roughly 10% of the mean urban concentration. In some months, such as August, the mean difference is negligible and in other months, for example, September, the mean difference is › 5µg/m³, or 22%. The fact that regional background levels can account for almost 90% of the PM_{10} observed in an urban area does contrast strongly with common views on the relative importance of local vehicular sources (both from exhaust and re-suspension).

Comparisons of monitoring data collected within Norwich itself show even less difference (note this was for a later period to that presented in Figure 3). The city council's mobile monitoring unit (purchased following a successful SCA application) was located at three different roadside/kerbside sites over a two year period (from March 1999 to March 2001). The scatterplot (Figure 4) shows that on many days PM_{10} levels at Norwich Centre were higher than at the chosen roadsides, in fact for 7 out of the 24 months, monthly average PM_{10} levels were greater at the Norwich Centre site than at the mobile. Over the whole 2 year period the mean difference between the two sites was less than 1µg/m³, varying between 0.0µg/m³ for the Bracondale site and 1.4µg/m³ at St. Stephens Street (the closest of the three sites to a road and with the highest proportion of taxis and buses). Although daily mean PM_{10} levels show a great degree of variation with the mobile unit recording levels between 86% greater and 80% less than Norwich Centre, the occasions in October 1999 and March 2000 with outliers on the scatterplot are thought to be related to roadwork activities taking place during this period rather than any direct traffic effect.

MODELLING WITH NAME

The NAME modelling was carried out for primary particles, ammonium sulphate and the sulphate component of other sulphate species. Emissions were input for an area covering most of Europe and the model was run using meteorological data fields from September 1997 to December 1998. PM_{10} output was modelled for nine sites across the UK where PM_{10} monitoring takes (Belfast, Bolton, Leamington Spa, Lough Navar, Narberth, Norwich Centre, Reading, Redcar and Rochester) and modelled SO_4 was compared with monitored data from the eight stations of the rural sulphur network. The model was set up and run so that source attribution was divided into five categories, four representing primary PM_{10} derived from East Anglian traffic sources, other East Anglian sources, the rest of the UK, and the European mainland (and Eire), and another representing sulphate (divided into both ammonium sulphate and other sulphate).

Whilst the average Mean Bias Error of the model for sulphate aerosol was -0.70µg/m³ (36% of mean observed levels) for PM_{10} this rose to between 8 and 18 µg/m³ (72 – 86% of observed levels) for the nine sites across the UK. Whilst the MBE of the sulphate aerosol predictions was greatest the further south-east the receptor, the error in the PM_{10} prediction was greatest at Urban Centre locations and then decreased as the sites progressed towards rural (e.g. Urban Centre, Urban Background, Suburban, Rural). In all cases the model underpredicted observed concentrations.

The reason for the difference in model accuracy can be explained by a number of factors. Firstly, the emissions data (taken from the National Atmospheric Emissions Inventory and the EMEP programme) does not include all sources of PM_{10}. Missing sources include not just windblown dust, sea-salt and resuspended road-dusts, but also small industrial (Part B) processes and minor roads. Secondly the use of gridded emissions inputs (2km resolution for East Anglia, 10km for the rest of the UK and 50km for the rest of Europe) means that where the receptor points are close to an emissions source, then the disaggregation of those emissions to a much larger area source will dilute the modelled concentration. As a starting point for estimating the contribution of the coarse component not included in the model, a figure of 10µg/m³ was used based on advice given in the NETCEN LAQM webpages (http://www.aeat.co.uk/netcen/airqual/laqm/pm10_sec/pm2top.html). This figure was later found to be questionable due to the fact that it was arrived at through the application of the 1.3 TEOM/gravimetric correction factor to a figure of 8µg/m³ derived by APEG.

Once the coarse component was added to the total modelled PM_{10} a factor of 2.45 (taken from APEG) was applied to the sulphate component in order to approximate total secondary aerosol. The final results were then compared to observed concentrations (see Figure 5). The most noticeable outcome of the comparison was that the adapted model results still under-predicted concentrations at all receptors except the three rural sites. This suggests (as posited above) that the 10µg/m³ may be too high as a general background figure, especially considering the other components absent from the modelling. It may be, however, that this figure is just too high for rural environments but might be more appropriate for urban areas where local anthropogenic activity leads to increased levels of coarse particles (for example due to re-suspension by traffic activity). If this is the case, then some distinction needs to be made between 'natural' and anthropogenic sources of coarse material so that the potential for management of the latter can be duly assessed.

Any remaining difference between predicted and observed concentrations in urban areas is likely to be attributable in part to the problem of sources being disaggregated over a large grid (see above). This has been necessary due to the constraints of computing power (even with a state-of-the-art PC the model runs used over 1-year of processor time). In order to assess how much of an effect this had ADMS-Urban was used to model the contribution to primary PM_{10} from road transport in Norwich.

MODELLING WITH ADMS-URBAN

Due to the impracticalities of validating a model such as ADMS-Urban for PM_{10} the model was validated against a network of nitrogen dioxide diffusion tubes operated by Norwich City Council. After setting up the model to produce satisfactory results for NO_2 from the road network for 1996, it was then adjusted to predict PM_{10} levels from these sources for 1998 (after having adjusted traffic figures). The road network used consisted of all the major (A, B and some other) roads in a 12x12km grid covering the city. The outcome of the modelling was a prediction of a 3.97μg/m^3 annual average contribution to PM_{10} concentrations at Norwich Centre AURN site. This compared with a prediction by NAME of only 0.18μg/m^3 from East Anglian traffic sources. The cause of this could be due to the use of disaggregated emissions in the NAME runs, the use of point receptors in ADMS-Urban (as opposed to grid) or the calculation of emissions using a bottom-up methodology for the explicit modelling of roads leading to higher emissions in the ADMS-Urban modelling. With the supposition made in APEG that exhaust emissions may only constitute 50% of vehicle derived PM_{10}, this may lead to up to 8μg/m^3 of the annual average being attributable to road transport compared with an annual average PM_{10} level at Norwich Centre AURN in 1998 of 20μg/m^3 and this is at what is effectively an Urban Background site.

LINKING THE RESULTS

Table 1 shows the results of the two models combined by replacing the East Anglian traffic contribution predicted by NAME with the ADMS-Urban prediction for the contribution to primary PM_{10}. The prediction for sulphate from NAME has been multiplied by the APEG factor of 2.45 to give a value for total secondary particles (as measured by TEOMs).

The modelled PM_{10} accounts for just over half of observed concentrations, with an unaccounted for difference of just over 9μg/m^3. This final portion is likely to consist of a combination of an additional contribution from non-traffic PM_{10} sources (as these are likely to have been underestimated both due to the grid problem and low quality emissions data) and a coarse component. It was described earlier how the coarse component found at urban background locations is likely to be greater than that at rural sites suggesting increased levels due to anthropogenic activities such as traffic re-suspension. This is supported by chemical analyses carried out in the DTI/ETSU study, which found that PM_{10} samples from Norwich contained more shale/soil/quartz than those at Stoke Ferry.

CONCLUSIONS

This study has shown clearly the need to consider both urban and region scales when modelling PM_{10}. The results suggest that only 20% of primary particles at the Norwich Centre site are attributable to vehicle related emissions. The low level of difference in the total PM_{10} observed between this site, roadside sites and the rural site suggests that PM_{10} is possibly very pervasive and doesn't share the same rapid decrease away from roads that has been shown to occur with NO_2. If this is true of re-suspended dusts as well as exhaust related particles then the scope for managing PM_{10} concentrations by reducing traffic rather than simply re-routing it or relying on national measures (cleaner fuels/engines etc) must be considered very carefully.

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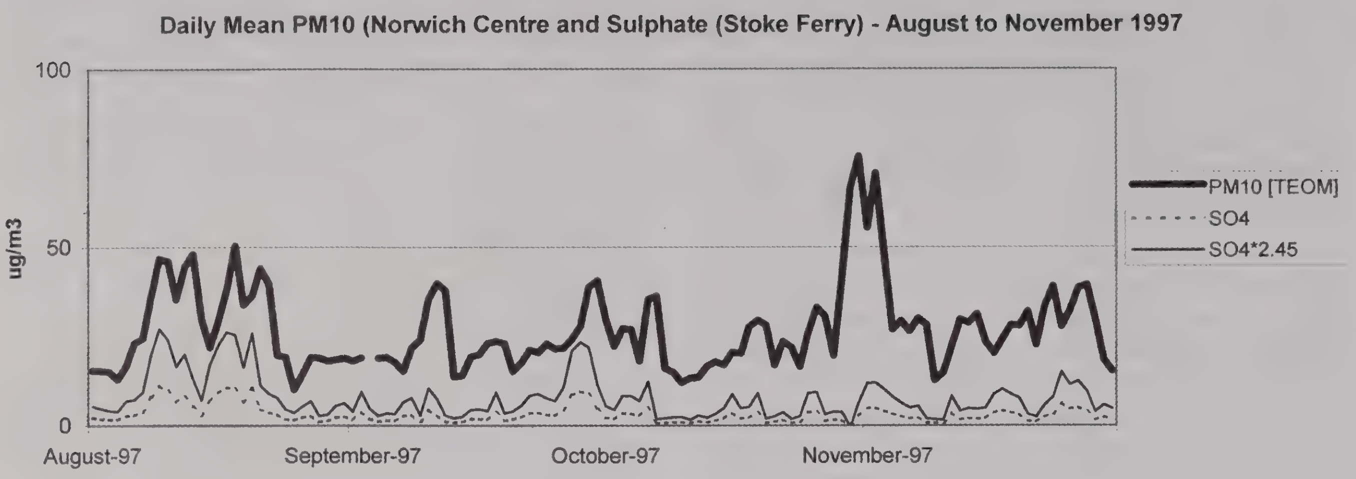
Figure 1.



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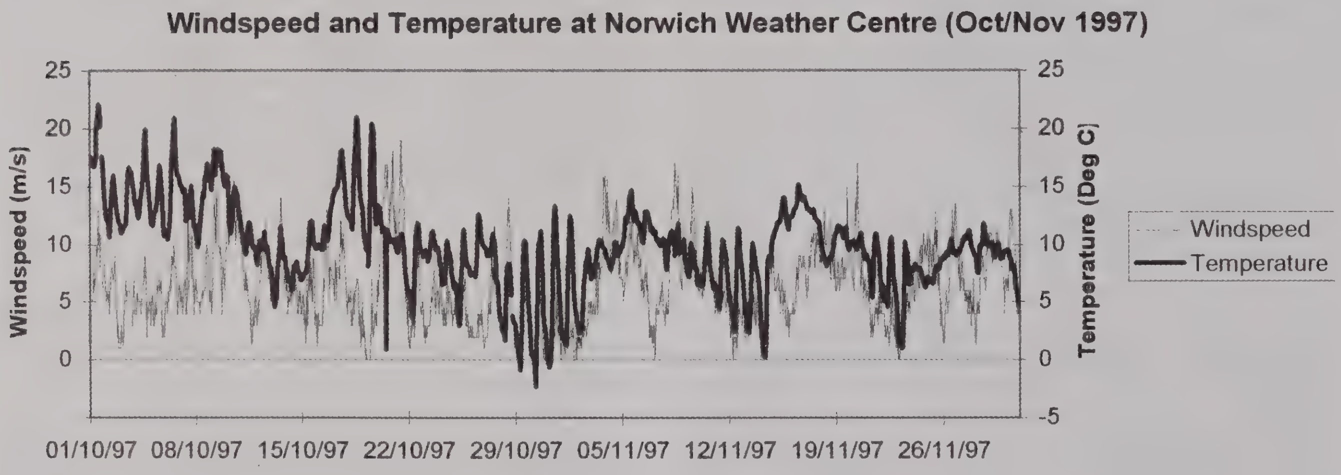

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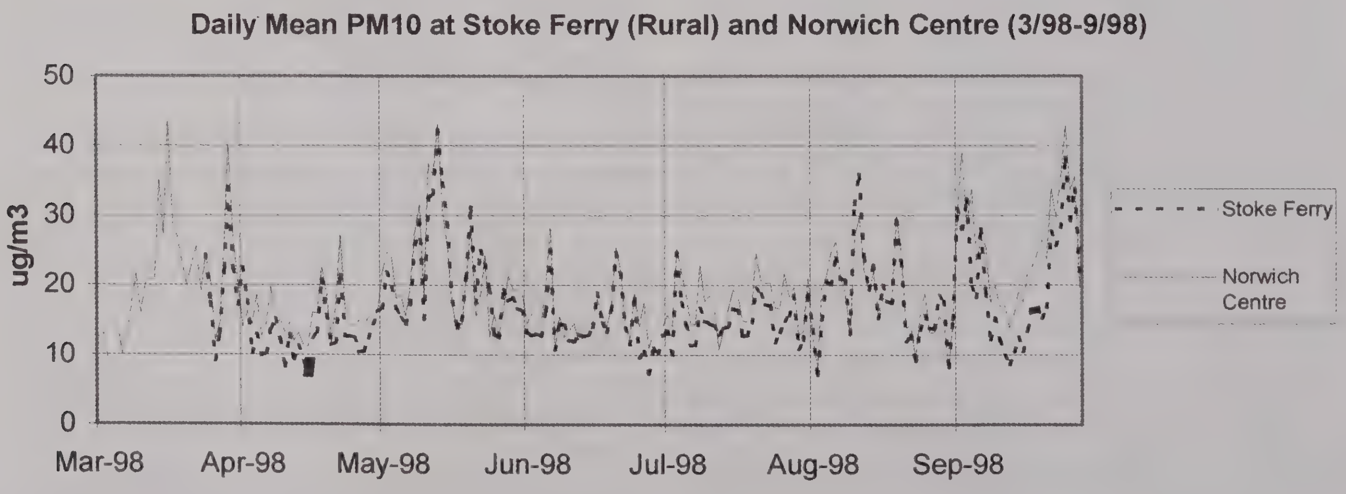

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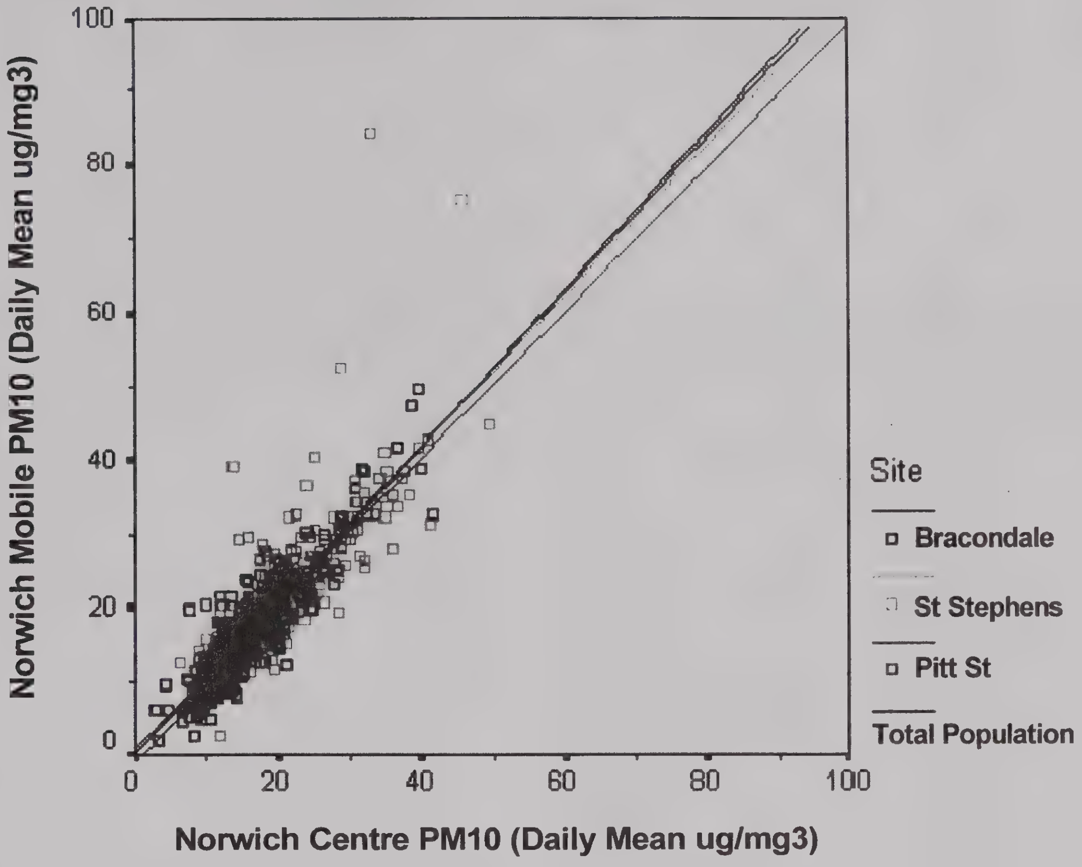

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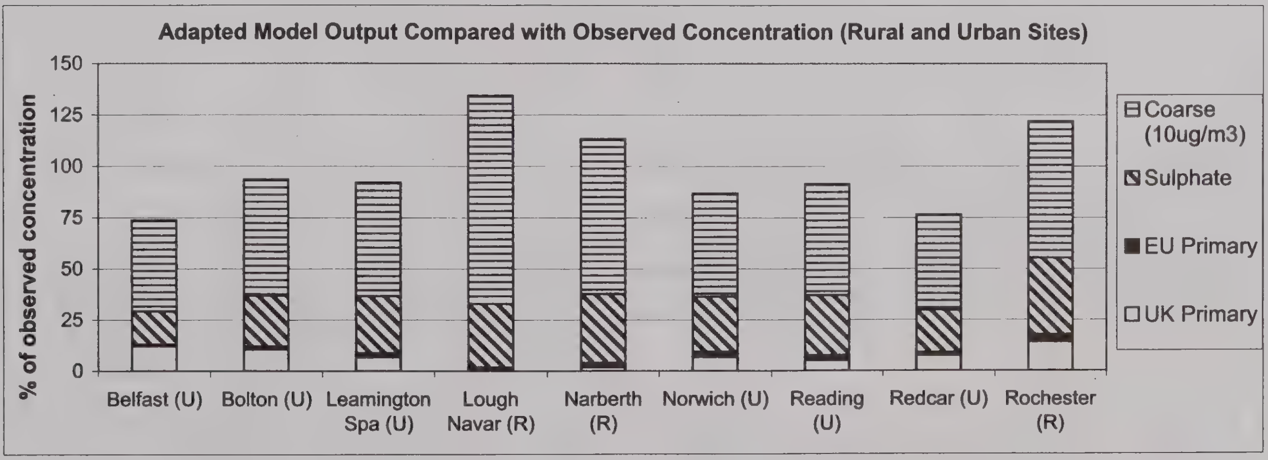

Factors Influencing Indoor Air Quality – Quantifying the Roles of Domestic and Outdoor Sources of Air Pollutants

Steven Naylor

Westlakes Scientific Consulting

ABSTRACT

Whereas air pollution is routinely monitored outdoors as part of obligations under the National Air Quality Strategy, much less is known about levels of air pollution indoors. A study was carried out in Whitehaven, West Cumbria in July 2001, in order to quantify short-term variation in domestic concentrations of nitrogen dioxide and to identify likely factors determining observed trends. Nitrogen dioxide levels were measured by chemiluminescence typically over a 24-hour period in a number of houses. The levels of nitrogen dioxide recorded indoors were compared to the standards for ambient outdoor air set out in the National Air Quality Strategy. During the period of indoor monitoring in the study, typical background concentrations in the centre of Whitehaven were between 10 and 15 ppb, and rose to between 20 and 25 ppb during the morning and evening rush of traffic. The results for indoors clearly indicate that when gas appliances are in use in a home, the nitrogen dioxide produced contributes overwhelmingly to indoor concentrations. Use of a gas cooker added around 100 ppb to background concentrations of nitrogen dioxide in the kitchen. Even a vented source such as a gas boiler, when actively heating water, added around 10 ppb. In contrast, indoor concentrations appeared largely unaffected by sporadic increases and decreases in concentrations outdoors, such as those occurring during the morning and evening rush of traffic. A 1-hour mean concentration of 133 ppb was recorded in the kitchen of one of the houses, which substantially exceeds the NAQS standard of 105 ppb. High occupancy groups, such as the elderly, may be particularly at risk to high exposure to nitrogen dioxide indoors and it would seem prudent to carry out further work in order to attempt to quantify potential public health implications.

INTRODUCTION

The National Air Quality Strategy sets out benchmark standards for ambient air with respect to levels of key air pollutants to assist local authorities in the management of air quality. The ultimate objective of the Strategy is to ensure that exposure of the general population, including susceptible groups such as the elderly, to the key air pollutants is kept sufficiently low so as to avoid ill-health. The primary emphasis of the National Air Quality Strategy is on outdoor air quality. However, the indoor environment is unable to free its occupiers from the adverse effects of air pollution. Whilst the low ventilation rates, typical of modern, well-insulated buildings, reduce pollution of indoor air caused by outdoor pollutants, they also inevitably increase exposure to air pollutants from indoor sources. Indoor sources of air pollution are numerous and include gas appliances, open fires, tobacco smoke, building materials, furnishings and pets. In homes with such sources present, levels of certain air pollutants indoors may periodically exceed those outdoors. Levels indoors may also periodically exceed the standards for the air pollutants set in the National Air Quality Strategy.

BACKGROUND

Whereas air pollution is routinely monitored outdoors as part of obligations under the National Air Quality Strategy, much less is known about levels of air pollution indoors. A study was carried out in Whitehaven, West Cumbria in July 2001, in order to quantify short-term variations in domestic concentrations of nitrogen dioxide and to identify likely factors determining observed trends. Nitrogen dioxide concentrations were measured continuously, typically over a 24-hour period, by chemiluminescence. The concentrations of nitrogen dioxide recorded indoors were compared to the standards for ambient outdoor air set out in the National Air Quality Strategy. The standards set out in the Strategy are based on current experimental and epidemiological evidence on the health effects of air pollutants. For nitrogen dioxide, two standards exist, a 1 hour mean standard of 105 ppb not to be exceeded more than 18 times a year, and an annual mean standard of 21 ppb.

The majority of studies to date that have measured ambient concentrations of nitrogen dioxide indoors have used Palmes diffusion tubes, which provide a measure typically averaged over several weeks. Such a measure is adequate for assessing the chronic health effects of long term exposures and is particularly suited to use in epidemiological studies, which generally require large sample sizes. However, diffusion tubes, because of the passive manner in which they operate, are unable to provide a reliable measure of nitrogen dioxide concentrations averaged over hours, which may be a more relevant period over which to measure exposure. The most widely used active method of measuring ambient concentrations of nitrogen dioxide is by chemiluminescence. Chemiluminescent analysers are able to provide high-resolution (typically down to 1 ppb) minute averaged nitrogen dioxide data.

METHODOLOGY

A number of houses were visited in July 2001 in order to measure domestic concentrations of nitrogen dioxide. The geographical focus of the study was the town of Whitehaven in the Borough of Copeland. Earlier in 2001, Copeland Borough Council established a permanent air quality monitoring station in the centre of Whitehaven, which provides data on ambient outdoor concentrations of nitrogen dioxide. Such data was made available to the study by the Council for the period over which indoor monitoring took place. The monitoring station is located approximately 25m from a main road carrying the bulk of traffic to and from the town centre. Nitrogen dioxide concentrations are measured at the monitoring station by chemiluminescence. Households were eligible for recruitment to the study if they lived in close proximity (i.e. less than 500m) to the monitoring station. A number of eligible households were invited to participate in the study and of those that agreed, a sample who reported using gas appliances were visited. During the visit, nitrogen dioxide concentrations were measured continuously in the kitchen, typically over a 24-hour period. In addition, information on domestic factors influencing indoor concentrations of nitrogen dioxide was collected by questionnaire. Nitrogen dioxide concentrations were measured using a chemiluminescent analyser deployed at the kitchen entrance such that the air inlet was away from draughts and at head height.

RESULTS

During the period of monitoring in the study, typical background concentrations in the centre of Whitehaven were between 10 and 15 ppb and rose to between 20 and 25 ppb during the morning and evening rush of traffic. Snapshots of minute averaged nitrogen dioxide data are presented in Figures 1 to 5 for two houses, one visited on 18 and 19 July, and the other on 25 and 26 July, 2001. Both dwellings were terraced houses built pre-1919 with a gas cooker and gas water boiler in the kitchen.

During monitoring on the 19th, the only indoor source of nitrogen dioxide was the gas water boiler, which was actively heating water for a short period in the morning between 08:50 and 09:00. The resulting rise in nitrogen dioxide concentrations in the kitchen during this period is shown in Figure 1. Concentrations start to rise from a background of around 18 ppb at 08:48, reach a peak of 25 ppb by 08:54, then gradually fall over the next half hour as the nitrogen dioxide produced by the boiler disperses. Background concentrations are returned to by around 09:30. The kitchen window was open between 09:45 and 22:30, hence the relatively short period between the peak nitrogen dioxide concentration and the return to background. For the most part of the rest of the day, nitrogen dioxide concentrations fluctuate narrowly between 13 and 17 ppb, closely mirroring the more widely fluctuating outdoor concentrations. The morning and evening peaks in traffic are apparent in the outdoor data, and the decline in indoor concentrations from around 16 to 12 ppb between 17:00 and 19:00, more than likely reflects the end of the evening traffic peak.

During monitoring on the 25th, the gas hob and grill was used for cooking between 17:15 and 17:30, whilst the gas boiler was actively heating water between 18:00 and 22.30. The kitchen window was open between 17:15 and 17:50. Their effects on nitrogen dioxide concentrations in the kitchen are illustrated in Figures 2 and 3. The rise in concentrations on the onset of cooking is clearly evident in Figure 2. Concentrations peak at 17:23 at 138 ppb then, with cooking over and the kitchen window open, immediately and rapidly decrease and have returned to background by 17:50. The effect of the active water boiler on nitrogen dioxide concentrations is shown in Figure 3. On the onset of water heating at 18:00, concentrations rise from a background of around 12 ppb and fluctuate around 25 ppb until 22:30. However, with the kitchen window closed, it is not until around 00:30 that concentrations have returned to background.

During monitoring on the 26th, the gas oven was used for cooking between 16:00 and 17:30, whilst the gas boiler was again actively heating water between 18:00 and 22.30. The kitchen window was open between 14:30 and 18:00. Their effects on nitrogen dioxide concentrations in the kitchen are illustrated in Figures 4 and 5. Figure 4 shows consistently high levels between 16:00 and 17:30, coinciding with the prolonged use of the oven. A peak of 200 ppb is reached at 16:17, although concentrations tend to fluctuate around 130 ppb for the duration of cooking. With the window open and the kitchen well ventilated, concentrations have returned to background by around 17:40. The effect of the active water boiler on nitrogen dioxide concentrations on the 26th is shown in Figure 5. Trends follow a similar pattern to those observed on the 25th, with around 12 ppb added to background during heating between 18:00 and 22.30. However, the decline to background after 22.30 is much quicker than on the previous day, with a concentration of 12 ppb returned to by around 23.00.

DISCUSSION

The indoor monitoring campaign carried out as part of this study allows the contribution to indoor nitrogen dioxide concentrations of different gas appliances in the home to be quantified. In the absence of active sources of nitrogen dioxide in the home, indoor concentrations tend to equilibrate to background outdoor concentrations. During the period of monitoring in the study, typical background concentrations in the centre of Whitehaven were between 10 and 15 ppb. The results for indoors clearly indicate that when gas appliances are in use in a home, the nitrogen dioxide produced contributes overwhelmingly to indoor concentrations. Use of a gas cooker added around 100 ppb to background concentrations of nitrogen dioxide in the kitchen. Even a vented source such as a gas boiler, when actively heating water, added around 10 ppb. In contrast, indoor concentrations appeared largely unaffected by sporadic increases and decreases in concentrations outdoors, such as those occurring during the morning and evening traffic rush.

The persistence of elevated concentrations of nitrogen dioxide indoors is very much influenced by the amount of room ventilation. This is supported by study results since simple measures such as the opening of windows while a source is active, appear very effective in quickly dispersing the nitrogen dioxide produced. For example, with kitchen windows open, peak concentrations resulting from cooking quickly returned to background, typically within half an hour. Extractor fans positioned directly over gas cookers would improve kitchen ventilation further and should be an industry norm. In addition, periodic checks of vents associated with vented gas appliances and boilers would ensure indoor build up of nitrogen dioxide due to their use is kept to a minimum and should be more widely recommended.

Based on the results of monitoring, it appears likely that the National Air Quality Strategy 1-hour mean standard for nitrogen dioxide is exceeded in a gas-fired kitchen far more than the permitted 18 times per year. Figure 4 illustrates the mean concentration for the period 16:00 to 17:00 on the 26th, coinciding with the use of a gas oven, and its relation to the National Air Quality Strategy 1-hour mean standard. The mean concentration for the period 16:00 to 17:00 was 133 ppb, which substantially exceeds the NAQS standard of 105 ppb. The significance of the nitrogen dioxide concentrations recorded over the period in terms of a person's personal exposure is beyond the scope of the study. However, considering the study was carried out in the summer when gas space heating appliances were not in use and rooms were well ventilated, there would appear a potential for clinically significant periodic rises in concentrations of nitrogen dioxide. It may be that these are large enough to impact even in the more frequently lived-in parts of the house, particularly in the winter months when gas heaters may be in use and rooms are generally less well ventilated. High occupancy groups, such as the elderly, may be particularly at risk to high exposure to nitrogen dioxide indoors and it would seem prudent to carry out further work in order to attempt to quantify potential public health implications.

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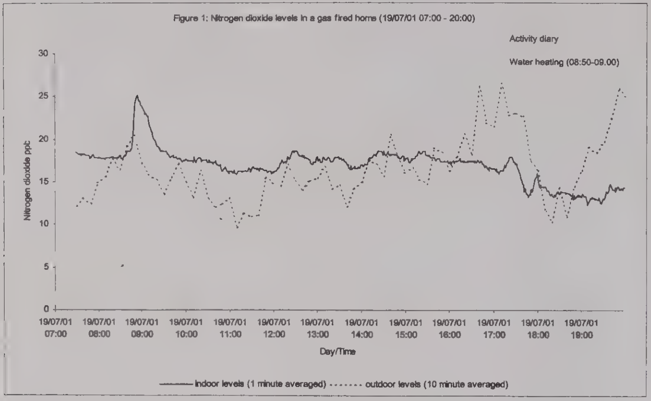
Figure 1: Nitrogen dioxide levels in a gas fired home (19/07/01 07:00 - 20:00)

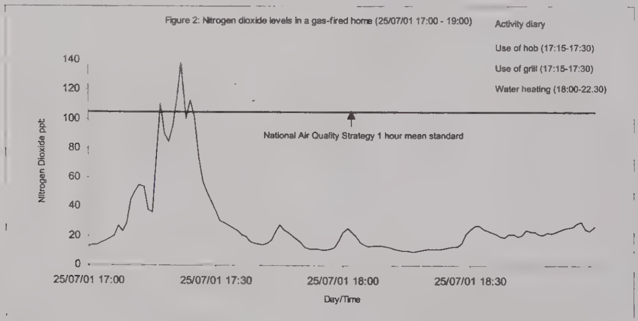
Figure 2: Nitrogen dioxide levels in a gas-fired home (25/07/01 17:00 - 19:00)

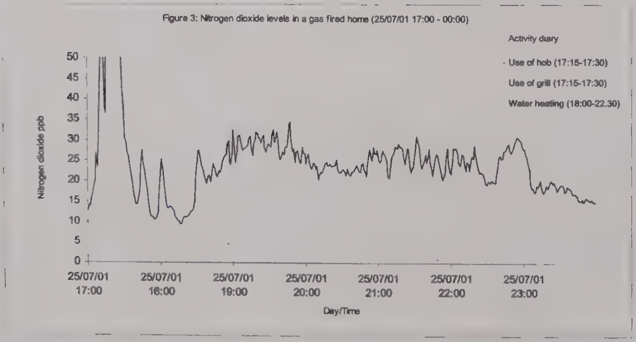
Figure 3: Nitrogen dioxide levels in a gas fired home (25/07/01 17:00 - 00:00)

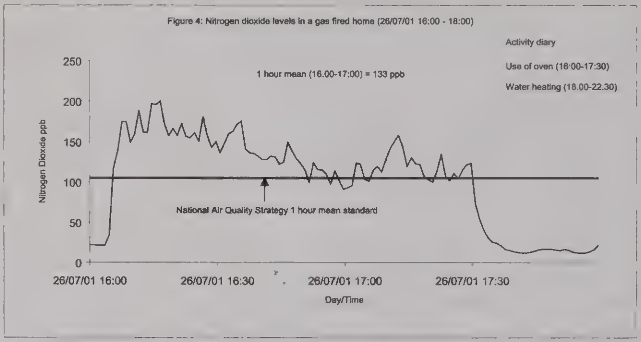
Figure 4: Nitrogen dioxide levels in a gas fired home (26/07/01 16:00 - 18:00)

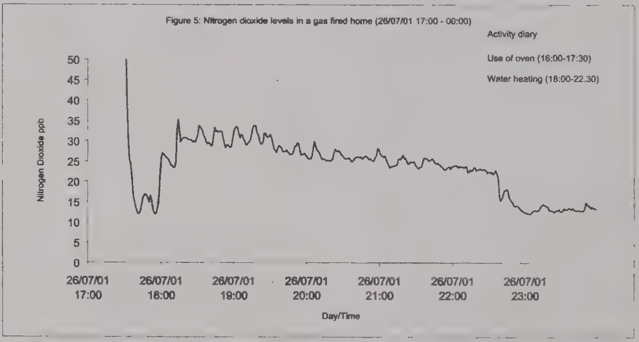
Figure 5: Nitrogen dioxide levels in a gas fired home (26/07/01 17:00 - 00:00)

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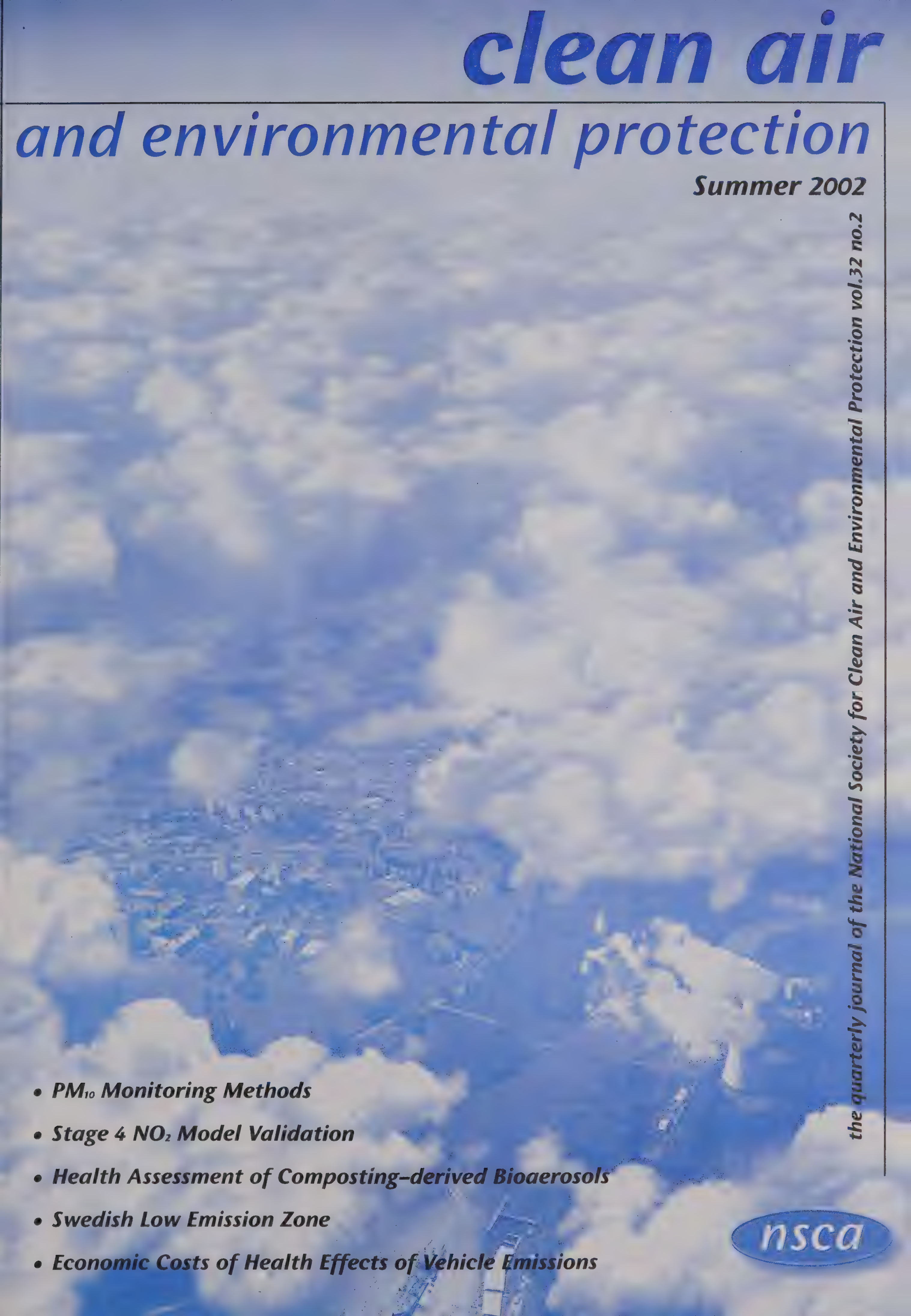

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CONTENTS

Editorial 35

PM_{10} Concentrations in the London Borough of Camden: Comparisons of Sampling Techniques 37
Anna Tod, Quentin Given, Caroline Scott

Stage 4 NO_2 Model Validation in Outer London 42
Alaric Lester, Steve Potter

Health Assessment of Composting-derived Bioaerosols 46
Martin Lake

Swedish Experience with Low Emission Zones 51
Christer Johansson, Lars Burman

Economic Costs of Health Effects of Vehicle Emissions 58
Alistair Hunt

The National Society for Clean Air and Environmental Protection produces information, organises conferences and training events, and campaigns on air pollution, noise and environmental protection issues. Founded in 1899, the Society's work on smoke control led to the Clean Air Acts. More recently NSCA has been influential in developing thinking on integrated pollution control, noise legislation, and air quality management.

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Editorial

AIR QUALITY: ATTACK OF THE CLICHÉS

The phrase "this is now the crucial period for local air quality management" has long since entered the NSCA cliché lexicon, and is vying for first place with "noise is the Cinderella pollutant" and "[pretty much everything] is a sustainable development issue". Nevertheless, this certainly is an important period for air quality, and for a number of differing reasons.

Most obviously, for those local authorities with Air Quality Management Areas, the pressure is on to complete Action Plans and have them implemented. A few are now emerging, but, as the NSCA Spring Workshop revealed, there is concern in Government and elsewhere that many could turn out to be "damp squibs". There are few, if any, areas of local government policy where the profession tasked with developing a plan has almost none of the tools at its disposal to carry it out. The crucial task here is to show that local authorities can deal with complex issues in an integrated way. In this issue, we carry a report from Sweden on experience there with Low Emission Zones, one of the possible tools available to meet the Action Plan challenge.

For those authorities without action plans, mid 2002 is crucial for different reasons. The next round of Review and Assessment is due for completion by the end of 2003, and revised guidance will be issued by DEFRA at the end of this year. For the majority of authorities, this means a 12 month gap between the end of the last review and the start of the next – plenty of time for budgets to be re-allocated, key staff to move on, and management commitment to wane. However, there will only be 12 months to complete the next review, unlike the 36 eventually allowed last time. This is therefore a time to be gearing up, not winding down, and developing local air quality strategies – as over 60% of authorities appear to be contemplating – could be a way of maintaining momentum.

On the national policy front, there are a number of key questions – will air quality drop off DTLR's radar? Can the Highways Agency be persuaded to be anything more than a cameo player in LAQM? Will air quality be sacrificed in the drive to "streamline" local government? In reality, the answers to these may well prove more positive than cynics would have us believe. However, there certainly could be a major shift in the focus of national air quality policy, away from the "urban" pollutants like NO_2 and particles, and towards "rural" pollutants, such as ozone and ammonia. With action on ozone and objectives for the protection of ecosystems among the strong candidates for the next part of the "rolling review" of the UK Air Quality Strategy, expect DEFRA to start showing its countryside face.

In this issue, we look at two of the thorny technical issues associated with air quality assessment – monitoring particles and model validation. Anna Tod reports on a comparison between different PM_{10} monitoring methods in Camden, and Alaric Lester shows how Croydon LB validated its ADMS Urban model for its "Stage 4" report. For those who wonder what their life is worth, we have the answer. Alistair Hunt, of Bath University, discusses some of the latest thinking in the estimation of the economic costs of air pollution, an important consideration for the Government in deciding how much further air quality objectives should go. Our fifth report, from Martin Lake at Exeter University, looks at the potential for airborne contamination from composting. The need to obtain permits from the Environment Agency for large composting schemes hit the newspapers recently, and Martin's report illustrates just why this is important.

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PM_{10} Concentrations in the London Borough of Camden: Comparison of Sampling Techniques

Anna Tod, Quentin Given, Caroline Scott

Pollution Projects Team, Environmental Health London Borough of Camden

1. Introduction and Background

The London Borough of Camden ranges from parts of Central London (including Covent Garden and Kings Cross) to Highgate and Hampstead Heath in the north and Kilburn in the west. As with the rest of Central London, road transport is the principle source of air pollution, accounting for over 70% of emissions.

Local authorities in the UK have a statutory requirement under Part IV of the Environment Act 1995 *and National Air Quality Strategy (DETR, 2000a) to review and assess air quality. Camden has completed Stages 1-3 of its air quality review and assessment and predicted that concentrations of PM_{10} or NO_2 would exceed Government health-based standards in 2004/2005 (see Table 1) in parts of the Borough. These areas were declared Air Quality Management Areas (AQMAs) in October 2000 on the basis of exceedances of either or both pollutants. Camden has continued to expand its monitoring strategy since its start in 1992 and on the basis of the review and assessment results has increased PM_{10} monitoring substantially.*

EU and national objectives are based on measurements by gravimetric techniques such as the Partisol where particulate matter is collected and weighed on filters to calculate a 24-hour average. However, as this technique is not automated, many local authorities use TEOM (Tapered Element Oscillating Microbalance) monitors to obtain real-time data. Much research has been carried out using both instruments and comparing results and it has been found that TEOMs tend to under-read when compared to gravimetric samplers, due to losses of semi-volatile aerosols from their heated element (e.g. Ayers et al, 1999). It is therefore recommended that raw data obtained from TEOMs are corrected by multiplying by 1.3 when comparing to the Government's health based standards (DETR, 2000c).

Camden Council has employed both methods to measure PM_{10} - concentrations in the Borough but also uses more low cost samplers. This paper highlights the different sampling methods that Camden used in 2000 and 2001 and compares the results. One of the reasons for comparing data is because of the large differences in instrument costs: it was hoped that if the cheaper, easy to operate and portable instruments produced comparable results to the more costly ones, then these could be used as a reliable alternative.

Table 1. National Air Quality Standards and Objectives for PM_{10} & NO_2

Pollutant	Health based standard		Date to be achieved
Particles (PM_{10})	*24h* 50g/μm^3 not to be exceeded more than 35/year	*Annual Mean* 40μg/m^3	31 December 2004
Nitrogen Dioxide (NO_2)	*1h* 105 ppb not to be exceeded more than 18/year	*Annual Mean* 21 ppb	31 December 2005

2. Monitoring in Camden

TEOM ambient particle monitors (from Rupprecht & Pashnick Co Ltd) have been used for fine particulate sampling at the Bloomsbury automatic station, an urban background site, since 1992. They have also been installed and operated in two Council owned automatic roadside sites – at Swiss Cottage ("Camden Roadside") from 1997 and at Shaftsbury Avenue (from 2000). Data from the first two sites are part of the Government's automatic network and are publicly available on Teletext, Ceefax, by freephone or on the Government's internet site under the automatic data section at www.aeat.co.uk/netcen/airqual. Shaftsbury Avenue data and Swiss Cottage data are presented on SEIPH's internet site at www.erg.kcl.ac.uk/home.asp.

Camden has one gravimetric sampler, a Partisol-Plus Sequential Air Sampler 2025 (Rupprecht & Pashnick Co Ltd), which was in operation during 2000 and 2001 at Camley Street Natural Park. As both Partisol and TEOM samplers are expensive and Partisols are also very labour intensive and do not produce real time results, Camden purchased several portable light scattering instruments (OSIRIS monitors from Turnkey Instruments). These were operated at various sites during 2000 and 2001, at roadside locations in the south of the borough (Brill Place and Macklin Street) and other sites near the Kings Cross development (York Way and Camley Street Natural Park) primarily for source apportionment and trend studies. OSIRIS monitors were also located at Swiss Cottage and near to the Shaftsbury Avenue automatic stations to compare against TEOMs. OSIRIS monitors can store several months' worth of data, which can then be downloaded via a computer modem link. In addition, at Camden Town Hall (an urban background site), the Council operated both a beta-ray absorption analyser (Beta Attenuation Monitor, BAM from ET Services Inc) and a size-fractionating Anderson Cascade Impactor, whilst another Impactor was run at Camley Street Natural Park for part of 2001. QA/QC procedures and data ratification for all techniques were carried out following LAQM guidance (DETR, 2000b); calibration of the majority of the instruments was regularly performed by outside contractors, and for the others by Camden Council itself. Locations of these particulate samplers are shown in Figure 1.

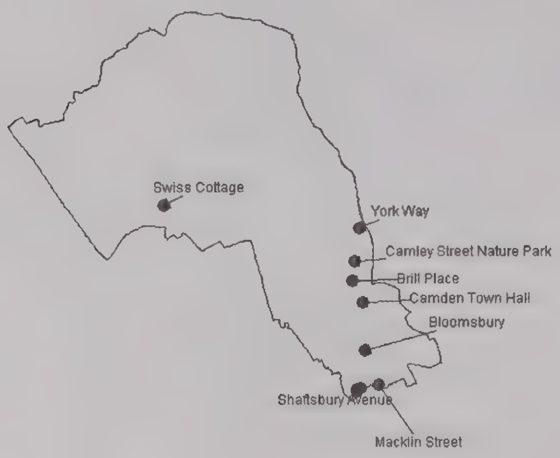

Figure 1. London Borough of Camden Particle Monitoring Sites in 2000 and 2001 The Swiss Cottage, Bloomsbury and Shaftsbury Avenue sites are automatic stations

Results are detailed in Section 3 for four major sites in the Borough and include comparisons between TEOM and OSIRIS data at Swiss Cottage and at Shaftsbury Avenue, Partisol, OSIRIS and Impactor data at Camley Street Natural Park and BAM and Impactor data from Camden Town Hall. TEOM data has been multiplied by the recommended factor of 1.3 so it can be compared to both OSIRIS data and to the national standards. The results are presented over short term intervals (24 hours) and longer term (weekly or annual averages) for a few sites. For example, this was needed when comparing data with results from the Impactor, where filters are only weighed weekly.

3. Results and Discussion

3.1 Comparison with National 24h Standards

The number of days when the national standard was exceeded in 2000 and 2001 at the various sites under investigation is shown in Table 2. Complete sets of data were not available at many sites for reasons including that they have not yet been ratified for 2001 (as for TEOM at Swiss Cottage), some instruments were only run for a certain period over the year (e.g. TEOM at Shaftsbury Avenue), and there may have been faults with instruments at some points. However if the data were to be extrapolated up to a year, results at most sites would be near to or greater than the national objective of 35 exceedance days.

Table 2. 24h PM_{10} Exceedances

	Number of days exceeded national standard of 50 µg/m$_3$	
Instrument, site	**2000**	**2001**
TEOM, Swiss Cottage	32	15 (6 months)
TEOM, Shaftsbury Ave	10 (4 months)	›35
OSIRIS, Camley Street	8 (4 months)	›35
Partisol, Camley Street	29	18 (6 months)
BAM, Camden Town Hall	›35	›35

3. 2. Instrument Comparison

3.2.1 Swiss Cottage

PM_{10} data from the automatic station at Swiss Cottage was collected from a TEOM and OSIRIS between August 2000 and June 2001. Figure 2 shows 24h results for a period of a month, during which time the data obtained from the two instruments correlated well, with the factored TEOM data generally being slightly higher than OSIRIS data (by a factor of 1.1-1.2). Both instruments exhibit the same daily peaks in PM_{10} concentrations which, during this sampling period, do not exceed the 24 hour national standard of 50µg/m^3.

When the data are compared over a longer time scale, such as over 7 days as in Figure 3, the correlation between PM_{10} concentrations from the two instruments is still good. However, unlike the 24h comparisons, concentrations averaged over 7 days did not show exactly the same trend over time. For example for the period of 8-23 April, OSIRIS results were greatest for the week of 9-15 April, whereas the highest concentrations from the TEOM were found in the week of 16-23 April. Similarly, as in Figure 2, results from the TEOM were higher than for the OSIRIS by a factor of approximately 1.2. There was one exception for the week of 12-18 February 2001, when a large increase in concentration was seen for both instruments, but for some reason, this is much more pronounced for the OSIRIS data, where concentrations exceeded the national 24h standard. This difference highlights a potential difficulty when comparing concentrations with a national standard as, if data from one instrument shows exceedances but another does not, it may be difficult to know how to report or interpret the data. It may therefore may be a good idea to compare several instruments over long time periods to investigate their reliability and accuracy.

3.2.2 Shaftsbury Avenue

As for the Swiss Cottage automatic station, OSIRIS and TEOM data were compared at Shaftsbury Avenue, a roadside monitoring station in the Clear Zone region in the south of the Borough. It should be noted that for this site, the two instruments are not in exactly the same place, being located across a major road. Despite this, the data still show good correlations and similar peaks in concentrations for the period between February and

May 2001 (Figure 4). Over this time, PM_{10} concentrations exceed the national standard on several days, and these exceedances are shown from both instruments.

3.2.3 Camley Street Natural Park

A portable OSIRIS monitor was compared with a gravimetric sampler (the Partisol) and a Cascade Impactor at Camley Street Natural Park, near to the Kings Cross development. Figures 5 and 6 show 24h results for OSIRIS and Partisol between January and March 2001 and Figure 7 shows a 7-day comparison between all three instruments. Similarly to comparisons with the TEOM analyser, the OSIRIS also correlates well with the Partisol and Impactor results.

Figure 5 shows that concentrations were closely related with OSIRIS results being slightly greater (by a factor of up to 1.2). It can be seen that on 1, 13 and 14 February, data from the OSIRIS exceeded the national standard but Partisol data did not. As it is recommended that Partisols be used to compare to standards, it may be best in such a situation to treat exceedances shown by the OSIRIS with caution. Partisol and OSIRIS data are also shown as a 1:1 comparison in Figure 6 where the majority of the data points lie near to the perfect fit line (Partisol = OSIRIS) and all but one result lies within the outer 2:1 lines. This one outlier is also seen in Figure 5 on 11 March 01, where the Partisol result is 1.7 $\mu g/m^3$ so is likely to be an error. If a best-fit trend line was drawn on Figure 6, it would have the r^2 value of 0.76.

In Figure 7, data have been averaged over 7 days in order for comparisons to be made with the Cascade Impactor. Data from all three instruments show the same trend over the time period, with peaks coinciding in the same week. Generally, data from the Cascade Impactor were higher than from the other two instruments by a factor of up to 1.3.

3.2.4 Camden Town Hall

The final comparison made between PM_{10} sampling instruments was between the Cascade Impactor and the Beta-Attenuation Monitor (BAM) at Camden Town Hall. It is more difficult to compare these two instruments as they produce results over very different timescales, BAM giving hourly averages and the Impactor giving weekly results for various particle sizes. However, if BAM data are averaged over the same timescale as Impactor PM_{10} data as in Figure 8, the results still show similar variations and concentrations over the period. For the first few months, the BAM tends to over-read (by a factor of 1.2), which is most likely due to its non-heated inlet which results in collection of water vapour. However in July and August, Impactor data is higher by a factor of approximately 1.3. There are likely to be more errors associated with Impactor data, which may be introduced from procedures such as handling and weighing filters. As with the other comparisons, there are incidences where data from one instrument gives exceedances of the 24h standard and the other does not.

Table 3. Annual PM_{10} Averages, Camley Street Natural Park

Year	Concentration ($\mu g/m^3$)		
	Partisol	OSIRIS	S.D
2000	29.59	26.32	2.31
2001 (6 months)	30.15	33.5	2.36

3.3 Long Term Averages

Results from the instruments used in this study were averaged over longer time scales to compare to the annual average standard. This was more difficult to do as most instruments have had gaps in sampling over the two years. An example of such results can be seen for Camley Street Natural Park (Table 3) for 2000 and 2001 (6 months only). Standard deviations of these averages were low for both years.

4. Conclusions

The London Borough of Camden has a range of particulate sampling instruments, including a gravimetric sampler (Partisol), automatic TEOM monitor, several portable light scattering OSIRIS monitors, a Beta Attenuation Monitor (BAM) and two Cascade Impactors. These instruments are normally used for different purposes depending on their cost, ease of use and type of data collected. This study has attempted to compare some of the data collected from these instruments at four different sites in the Borough over 2000 and 2001.

Similarly to other studies that have compared data from these and other particulate sampling instruments (Salter & Parsons, 1999, Chung et al, 2001), the results in this study have shown that both short term and long term comparisons between the instruments are consistent at all sites. Data constantly exhibits the same patterns over time and concentrations within a factor of approximately 1.3. This comparison has therefore been very useful for Camden Council as it has confirmed that instruments such as the OSIRIS, which can be easily moved around different sites in the Borough, can be used as an alternative to the TEOM, which is much more difficult to move around. Camden Council has purchased ten OSIRIS monitors, which are ideal for monitoring particles emitted from projects such as the Kings Cross development works. For example, almost half of the OSIRIS monitors are currently located at various sites around the development and these can be easily moved in order to monitor the PM_{10} concentrations as the demolition/construction works progress and traffic movement changes.

Comparing PM_{10} data obtained from the different instruments to the national health-based 24h standard of 50$\mu g/m^3$ needs to be treated with more caution. This is because the results have shown that on some days, data from one instrument may exceed this value, but data from another instrument for the same day may not. Therefore, when reporting the number of exceedance days, the sampling method used should be highlighted and in polluted areas where exceedances are expected, it may be best to run two instruments together to verify such data.

The instruments used in this study all use very different sampling methods to measure PM_{10} concentrations and all have operational advantages and disadvantages. They may also be most suitable for different purposes, such as for long term trends or size fractionating studies. Despite their differences, these comparisons have shown that the data obtained is very similar over the two years studied so it is suggested that any of these instruments could be used as reliable alternatives to sample PM_{10} concentrations.

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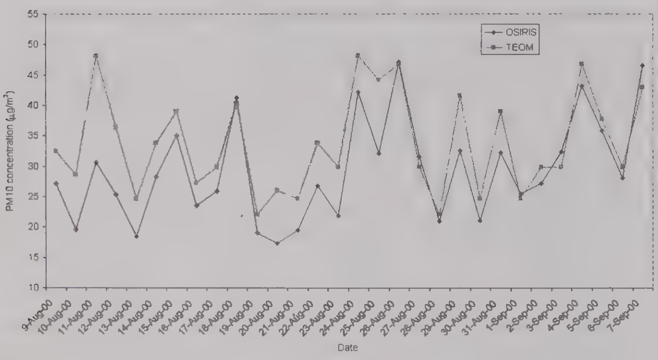

Figure 2. 24h PM_{10} Concentrations from OSIRIS and TEOM (factored), Swiss Cottage, Aug-Sept 2000

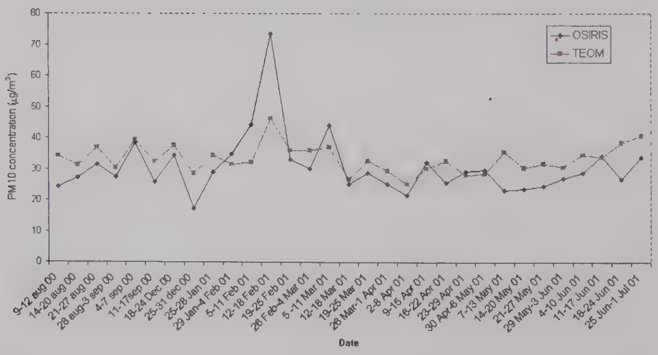

Figure 3. Seven day PM_{10} Concentrations from OSIRIS and TEOM (factored), Swiss Cottage, Aug 2000 – July 2001

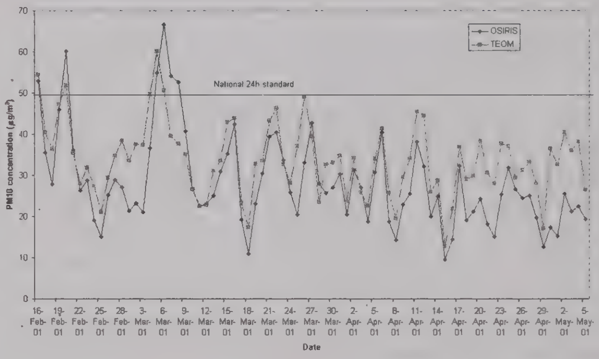

Figure 4. 24h PM_{10} Concentrations from OSIRIS and TEOM (factored), Shaftsbury Avenue, Feb-May 2001

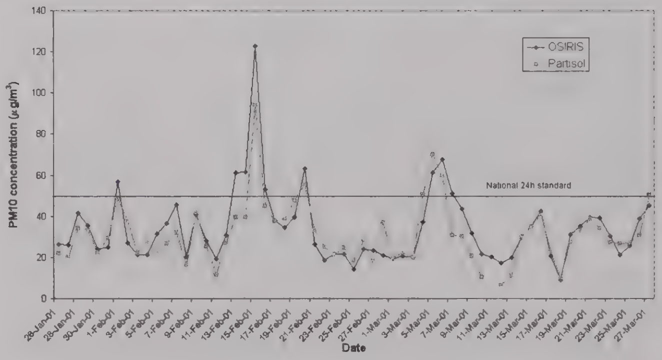

Figure 5. 24h PM_{10} Concentrations from Partisol and OSIRIS, Camley Street Natural Park, Jan-March 2001

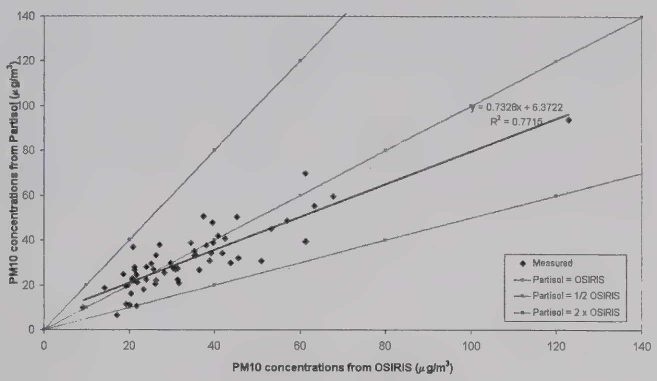

Figure 6. 1:1 Comparison of 24h PM$_{10}$ Concentrations from OSIRIS and Partisol, Camley Street, 2001

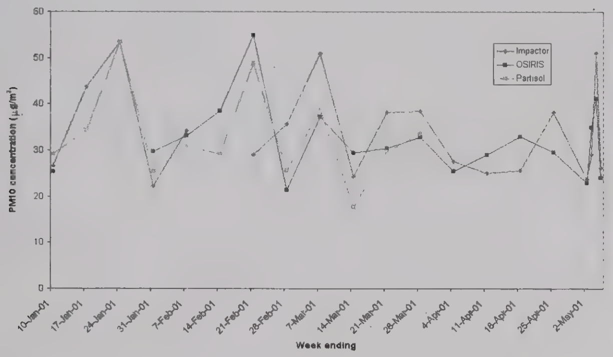

Figure 7. Seven Day Comparison of PM$_{10}$ Concentrations, Camley Street Natural Park, Jan-May 2001

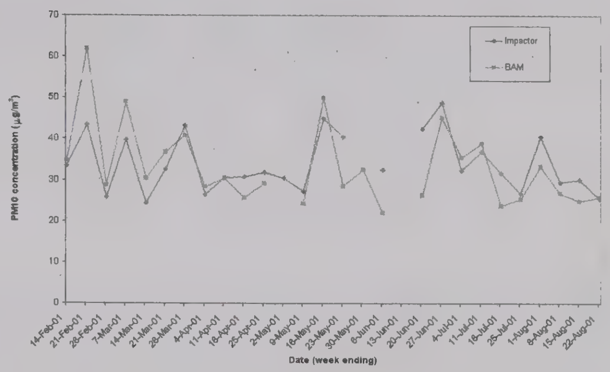

Figure 8. Seven Day PM$_{10}$ Concentrations from Cascade Impactor and BAM, Camden Town Hall, Feb-Aug 01

Stage 4 NO_2 Model Validation in Outer London

Alaric Lester, Steve Potter

London Borough of Croydon

Abstract

The London Borough of Croydon is situated to the south of London. Pollution levels are lower than in central London, but the nitrogen dioxide (NO_2) annual mean objective level is frequently exceeded at many of the borough's continuous monitoring sites. Croydon Council has expanded its continuous monitoring network to cover five sites. With three other sites close to the borough boundary and more years of historic data, there are significantly more data available for model validation than when the borough's air quality management area (AQMA) was originally declared. This report discusses validation of Croydon's stage 4 review and assessment work using ADMS Urban, issues arising and the implications for local air quality management.

Background

Croydon borough covers an area of approximately 86 square kilometres and has a population of 338,000, the highest of all London boroughs. It has a dense road network in the north and centre of the borough, with suburban and semi-rural areas to the south. Motor vehicles are the source of the majority of air pollution. The centre of Croydon has a number of tall buildings, and parts of the borough are also particularly hilly. These factors combine to give marked variations in air quality, with frequent exceedances of the annual mean objective level for nitrogen dioxide (NO_2) to the north and far lower pollution levels to the south (DETR 2000).

Reasons for Remodelling for the Stage 4 Review and Assessment

All local authorities that have declared air quality management areas (AQMAs) must undertake a stage 4 review and assessment. Since declaring an AQMA for NO_2, a number of factors have changed in Croydon:

- New emissions data
- An updated version of ADMS Urban, version 1.6
- New monitoring sites
- Met data from Heathrow, rather than Gatwick
- Queries from residents as to whether the AQMA was correct (some residents believed that the A23 in their area should have been declared originally).

As a result of these changes, the Council decided to remodel completely, and was able to validate against a far larger data set than was previously available.

Monitoring Sites

Three new continuous monitoring sites have been commissioned in Croydon since 1999, as well as one at Crystal Palace, just to the north of the borough, which Croydon Council runs jointly with adjacent boroughs. Monitoring site data from these sites, two Sutton sites and a site that has run since 1994 were used, with data from 1998 to 2001. Monitoring data for 2001 presented here are provisional and subject to change. However, final, ratified data are unlikely to be significantly different and 2001 results substantially increase the size of the data set. The sites are described below using their London Air Quality Network or Automated Urban Network (AUN) site codes.

- CR2 has been in operation since 1994. It is a roadside site on the A23, with open topography. Historically, the site shows low conversion of oxides of nitrogen (NO_x) to NO_2, more typical of a motorway site. This can be explained in part by the lack of significant emission sources in the vicinity other than the A23.
- CR4 is a roadside site in the centre of Croydon. It is situated at the junction of two roads with relatively low traffic flow, although a busy underpass is around 20 metres from the station.
- CR5 is also a roadside site on the A23. It is in a street canyon in the north of the borough, where significant traffic queuing occurs during busy periods.
- CR6 is classified as an industrial site, although the power station that it is set to monitor has not yet been commissioned. It is currently more representative of an urban background site.
- CY1 is the Crystal Palace monitoring station, just to the north of the borough. It is a roadside site, with a high density of parkland in the vicinity.
- Sutton 1 and 3 are AUN-affiliated sites in neighbouring Sutton. Sutton 1 is a roadside site, Sutton 3 urban background.

NO_x or NO_2 Validation?

Validation looks at the predictive performance of a package: input data (emissions, rural concentrations and site-specific information) and model combined. For the purposes of local authority review and assessment, the prime concern is whether the package performs satisfactorily for the purpose, that is, whether the package predicts NO_2 with reasonable accuracy. From a purely pragmatic point of view, when assessing against the NO_2 annual mean objective, a local authority is only interested in how well a model predicts annual mean NO_2. Also validating with NO_x, though, gives greater insight into true model performance. Since NO_x is effectively a primary pollutant, the effect of dispersion and of any chemistry effects can be seen separately. In this study, annual mean concentrations of NO_x and NO_2 were studied.

Model Set-Up

Emissions data were taken from the February 2002 release of the London Atmospheric Emissions Inventory (LAEI, GLA 2002), supplied by the Greater London Authority (GLA). The LAEI is a comprehensive database of emissions across London. The February 2002 version contains emissions for 1999 and 2005 and incorporates the updated emissions factors for NO_x and fine particles (PM_{10}) recently released by the Department for Transport, Local Government and the Regions (DTLR). Receptors were set up at the monitoring site locations above and all roads within 200 metres of these receptors were modelled explicitly. Where modelled road locations differed spatially from the actual roads, receptor co-ordinates were adjusted accordingly. Aggregated emissions for all sources within Greater London were included within 1km x 1km grids. For validation years other than 1999, the emissions data were scaled accordingly.

Meteorological data were taken from Heathrow, since Gatwick no longer records all the information required for input to ADMS Urban.

The chemistry module used was the generic reaction scheme, or GRS (CERC 1999). This requires emissions of NO_x and non-methane volatile organic compounds (NMVOC), as well as background NO_x, NO_2 and ozone. Background NO_x, NO_2 and ozone measurements were taken from rural stations around Greater London. These represent the pollution entering the region.

Site-specific parameters suitable for Croydon were chosen. A surface roughness length of 50 cm was used. Taking complex terrain into account, for instance, varying surface roughness across the borough would significantly increase computational time and not be appropriate for the timescale of the review and assessment process.

Results

Table 1 shows measured and modelled annual mean NO_2 data for all sites and all years, along with derived statistics (NO_2: NO_x ratios and the ratio of measurements to modelled results). Figures 1 and 2 show scatter plots of results. CR2 data are labelled as 'motorway' data.

Observations

NO_2 only

- Averaging over all sites and years, the model is accurate to within 5%.
- The model consistently over-predicts at CR2.
- The model under-predicts at CR5.

NO_x only

- The model consistently under-predicts at CR2, CR5 and CY1 (all roadside sites). At CR5, the model is out by more than 50%.
- The model consistently over-predicts at Sutton 3, a background site.
- Roadside sites tend to under-predict NO_x.

NO_x and NO_2

- Predictions are closer to measurements for NO_2 than NO_x.
- Predicted NO_2: NO_x ratios are more accurate at background sites.

Discussion

Looking at NO_2 results only, model results are generally good, with the exception of the CR2 and CR5 sites. An initial conclusion might be that emissions are over-estimated along the A23 by CR2 and under-estimated around CR5. As NO_x results show, though, this is clearly not the case at CR2.

Considering NO_x further, most roadside sites show under-predictions, with slight over-prediction at Sutton 3. Looking at NO_x alone, one might initially conclude that the LAEI under-predicts NO_x emissions from major roads in Croydon or that the model gives greater dilution than anticipated.

With a more varied topography, the dispersion of pollutants in outer London is different from, say, central London. In ADMS Urban, this can be accounted for by setting appropriate values for surface roughness, minimum value of Monin-Obukhov length and height of aggregated sources. (For the latter, ground-level emissions must first reach rooftop height before they can effectively be dispersed, so a higher value should be used in areas with tall buildings.)

Differing vehicle speeds may also affect turbulence. However, the initial height to which road emissions are mixed (and hence the initial dispersion) cannot be set by the user. Another factor of possible importance is the proportion of NO_x that is emitted directly as NO_2. In the absence of further information, the default value of 5% by volume was used. The DTLR research project UG294: TRAMAQ – Assessment of Primary NO_2 Emissions, is likely to conclude that a more realistic value is 10-12%, although the research has not been published at the time of writing this report.

It is worth considering chemistry and non-chemistry issues separately to understand the various mechanisms at work.

Non-Chemistry (emissions and dispersion for NO_x)

A number of explanations as to the pattern of results is possible. Some are discussed below.

NO_x emissions from roads may be under-estimates (or background NO_x under-estimated). The rural monitoring sites used to monitor background pollution may not be representative of actual pollution entering London. If emissions are under-estimated, it could be that there is greater variation in speeds than in the emissions calculations or that the vehicle fleet in south London does not match national profiles. However, if emissions on roads increase, modelled background concentrations would increase, since NO_x in London is primarily from road vehicles (GLA, 2002). This would not fit with background results from Sutton 3. The under-estimate is a likely factor at CR5, though, because of the degree of under-prediction. Indeed, there is significant traffic queuing by the site at peak-hours, which may not have been picked up by traffic and speed surveys.

NO_x road emissions (or background NO_x) may be under-estimated while the contribution of non-road sources is over-estimated. If the height of aggregated sources is set too low, this has the net effect of over-estimating the contribution from these sources. This would have a more significant effect at background sites. This cannot be ruled out.

Initial dispersal of roadside emissions in Croydon may be less than that assumed by the model, leading to lower predictions at roadside. This could be due to assumptions about initial mixing of road emissions. This would explain under-prediction of NO_x at roadside sites, compared to background predictions which are closer to measurements.

There are other possible problems at CR5. The topography may not have been adequately modelled (there is a railway bridge as well as a street canyon and the immediate area is at a low elevation).

Chemistry

The modelled NO_2: NO_x ratio is higher at most roadside sites than that observed, and significantly higher at CR2 and CR5. CR4, CR6 and Sutton 3 show good agreement. This is only part of the story, however. Figure 3 shows the proportion of NO_2 as a function of total NO_x. At high NO_x concentrations, the measured proportions at CR2 and CR5 show a significant spread.

As expected, the proportion of NO_2 decreases with total NO_x, as the oxidising potential of the air is reduced. Since roadside NO_x is under-estimated by the model, the proportion of NO_2 is consequently over-estimated. The net effect is to reduce any errors in NO_x predictions.

From a linear regression of the data in Figure 3, the model over-predicts the proportion of NO_2 by around 5% for any specified NO_x concentration.

Irrespective of these factors, CR2 remains a notoriously difficult site to model. Model results have consistently under-predicted NO_x and over-predicted NO_2.

The Local Air Quality Management (LAQM) Perspective

Areas such as CR2 may be declared unnecessarily within the review and assessment process, due to over-estimates of local NO_2 levels. From a precautionary point of view, this is of little concern. Of more concern are sites such as CR5, where the under-estimate in annual mean NO_2 is at least 20%. In these cases, the extent of the local pollution problem is likely to be under-estimated. The model predicts significant exceedances of the NO_2 objective level in any case, though in the predictive modelling for 2005, there could be areas identified as likely to be within the objective when this will not be the case. It may be that CR5 is an isolated case, but including a margin of error may be necessary when determining likely exceedance areas of the objective.

Because of its large area and widely varying topography and emissions profiles, an area such as Croydon is likely to be more difficult to model accurately than a more homogeneous area such as central London. There may, therefore, be a case for using a larger margin of error.

Conclusions

This study has shown that NO_2 can be modelled with a reasonable degree of accuracy within Croydon using ADMS Urban, although problems remain at sites such as CR2 and CR5. NO_x predictions have been less successful in Croydon, but without further study, the precise cause cannot be identified. Chemistry schemes should normally reduce NO_x errors in NO_2 predictions. In terms of review and assessment, a precautionary approach to declaration of AQMAs may be necessary in order to catch all exceedance areas where emissions may be exceptionally high or where site topography cannot be adequately described.

Further Work

With the next round of reviews and assessments on the horizon, and with the results of the TRAMAQ UG294 report, it would be useful to study further NO_x and NO_2 predictions in other areas. Sensitivity analysis may help to identify which factors cause discrepancies between model and measurements.

References

Cambridge Environmental Research Consultants (CERC) 1999, 'ADMS Urban User Guide'.

DETR 2000, 'The Air Quality Strategy for England, Scotland, Wales and Northern Ireland', The Stationery Office.

GLA 2002, London Atmospheric Emissions Inventory.

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Table 1: Results of Croydon Stage 4 Validation for NO_x and NO_2 Annual Mean Concentrations*

Site Type	Site Name and Year	Measured NO_x/ppb	Modelled NO_x/ppb	Measured NO_2/ppb	Modelled NO_2/ppb	Measured NO_2:NO_x	Measured: modelled NO_x	Modelled NO_2:NO_x	Measured: modelled NO_2
Mway	CR2 98	104.6	74.1	20.9	30.9	0.20	0.71	0.42	1.48
Mway	CR2 99	91.5	64.5	20.3	28.9	0.22	0.70	0.45	1.42
Mway	CR2 00	87.3	52.4	18.3	24.7	0.21	0.60	0.47	1.35
Mway	CR2 01	91.6	55.6	22.9	25.7	0.25	0.61	0.46	1.12
Roadside	CR5 00**	(114.6)	(54.9)	(29.0)	(23.1)	0.25	0.48	0.42	0.80
Roadside	CR5 01	128.0	59.1	37.7	27.1	0.29	0.46	0.46	0.72
Roadside	CY1 00	77.7	52.0	24.7	24.8	0.32	0.67	0.48	1.00
Roadside	CY1 01	80.9	55.5	25.5	25.7	0.32	0.69	0.46	1.01
Roadside	Sutt1 98	60.8	50.3	21.7	24.9	0.36	0.83	0.50	1.15
Roadside	Sutt1 99	60.6	64.8	22.6	29.4	0.37	1.07	0.45	1.30
Roadside	Sutt1 00	56.6	35.9	20.7	19.7	0.37	0.63	0.55	0.95
Roadside	Sutt1 01	61.1	41.5	23.1	21.4	0.38	0.68	0.52	0.93
Roadside	CR4 00	63.0	62.1	27.2	27.1	0.43	0.99	0.44	1.00
Roadside	CR4 01	70.9	63.7	29.8	28.0	0.42	0.90	0.44	0.94
Industrial	CR6 01	45.4	39.1	21.3	20.5	0.47	0.86	0.52	0.96
Background	Sutt3 98	34.0	45.4	17.5	23.7	0.51	1.34	0.52	1.35
Background	Sutt3 99	33.5	43.9	18	23.2	0.54	1.31	0.53	1.29
Background	Sutt3 00	29.2	32.5	16.9	18.4	0.58	1.11	0.57	1.09
Background	Sutt3 01	34.7	37.8	18.5	20.1	0.53	1.09	0.53	1.09
Average		**69.8**	**51.4**	**23.4**	**24.5**	**0.37**	**0.74**	**0.48**	**1.05**

* Data for 2001 are provisional and subject to ratification

** CR5 had data for only 3 months in 2000

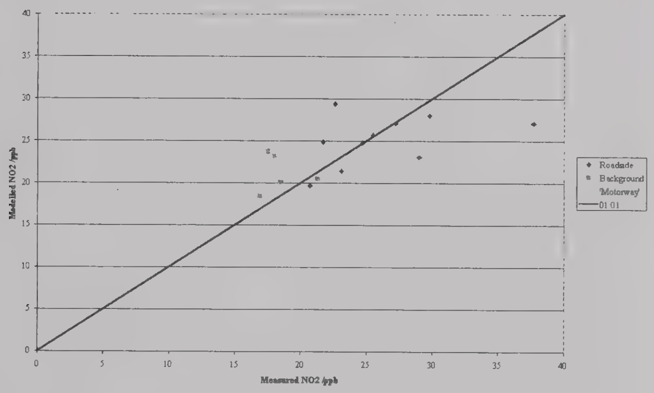


Figure 1. Modelled NO_2 versus Measured NO_2

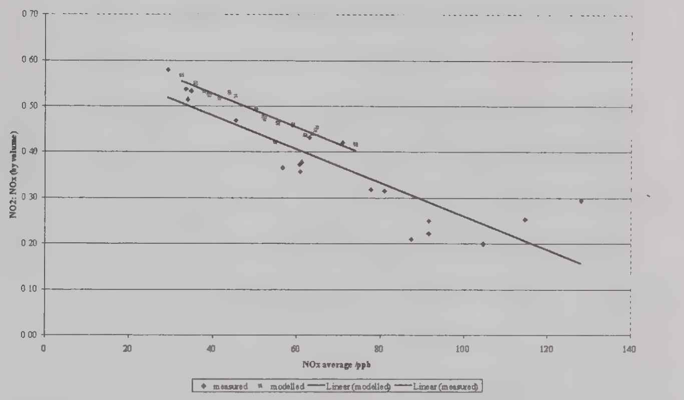


Figure 3. NO_2: NO_x ratio versus NO_x

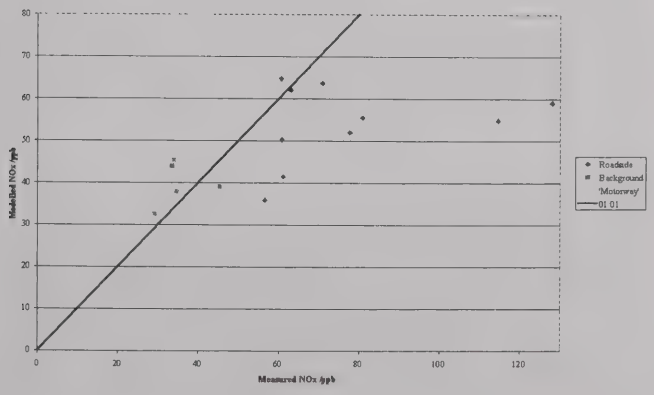


Figure 2. Modelled NO_2 versus Measured NO_x

Health Assessment of Composting-derived Bioaerosols

Martin Lake

Carrick District Council

An investigation into bioaerosols resulting from domestic composting has been conducted in Cornwall. Experiments employed Rotols and mimicked procedures and materials commonly utilised by the public, during such operations. They have shown that fungal spores such as Aspergillus Fumigatus, formerly associated with Farmer's lung, are produced in these systems. These and other species were readily sampled in the free-air space between the compost surface and the Rotol lid. Subsequent development of the spores on agar plates was readily achieved during the warmer summer months. All types of compost produced the same 'bioaerosol event'. Whilst fungal counts did not prove excessive, a precautionary approach is advised here for vulnerable groups. In this regard, a risk assessment flow-chart is proposed and aimed at members of the public who consider undertaking domestic composting.

Introduction

Composting of human and domestic waste is not a new phenomenon – the Chinese practised the process many centuries ago, and there are no records confirming that the process causes allergic or respiratory problems. Farming practices at the beginning of the 20th century, however, identified many agricultural workers at risk of developing *Aspergillosis,* Farmer's lung, or Allergic Bronchopulmonary *Aspergillosis.* This condition was caused by inhalation of *Aspergillus Fumigatus* fungal spores (1μm – 3μm in size) produced from mouldy hay, straw, and crushed cereals fed to livestock. Changes in farming practices, good health and safety programmes and the use of mechanical handling methods to reduce dust levels, have been a contributing factor in the reduction of incidences. Recorded numbers of *Aspergillosis* have fallen sharply to a level where the condition is not at all common. Little work has been carried out on production and dispersion of fungal spores from home composting, however, save from one or two investigations on large-scale systems.

Medical knowledge suggests that airborne particulates are causing increased allergic reactions and asthmatic attacks, which occur in all ages through the whole social economic spectrum. Against this, statistics from 1980 to 1998 are showing a drop in PM_{10} levels and an increase in days when pollution levels are below the medium to high level (NETCEN, 2000; DETR, 2000a). Nevertheless, it is estimated that in the UK, 1 in 10 children has suffered asthma and that 40% of the UK population will suffer asthma symptoms at some time by the time they reach the age of 33 (NSCA, 1997). Hypersensitivity of certain individuals is frequently associated with some adverse reaction and home composting might prove to be a triggering mechanism.

In an area such as Cornwall, very little difference has been registered in particulate levels between rural and urban areas (Salter & Parsons, 2000). Composting derived bioaerosols may only contribute a small fraction to the loading there but nevertheless, in areas such as this our understanding should benefit significantly from such study.

Background

Air quality monitoring internationally does not include standards for such pollutants. Consequently, the value of air sampling for them remains a relatively low level priority, though these organisms could present serious implications for health. The UK government's waste management targets and initiatives have identified through the waste hierarchy the valuable contribution composting can make towards reaching national and European targets (EU, 2000). A number of local authority and commercial composting schemes are well established. The commercial schemes will be monitored by the Environment Agency which has adopted the protocols for sampling and enumeration of airborne microorganisms produced (Composting Association, 1999). The former Department of Environment (DoE) through *Making Waste Work* (DoE, 1995) identified that 40% of households with gardens should be undertaking home composting by the year 2000. Local authorities embraced the targets by supplying free or subsidised home composting units for use by the public. However, the issues concerned with composting derived bioaerosols, especially the fungi *Aspergillus Fumigatus,* have received little investigation. Assessment of the relative safety of all groups involved in home composting activities therefore requires further research. As little scientific information is available for home composting systems in the UK, there is a specific need for a study to examine bioaerosol production and its release from a domestic composting system, where different types and or mixes of green waste are treated.

Composting

Household domestic waste analysis identifies approximately 25% of the stream as putrescible waste. During 1998/9 over 25 million tonnes – almost 480 kg per person – was collected by local authorities in England and Wales (CIPFA, 1999). Under the European Union Landfill Directive (EU, 2000) and the Waste Strategy for England and Wales (DETR, 1999; 2000b), the UK is committed to reducing biodegradable waste sent to landfill to 75% of the 1995 total by 2010.

The introduction of the *Control of Pollution Act* (1974) made sweeping changes to landfill operation, which were further reinforced and extended and the greater part repealed by the *Environmental Protection Act* (1990). Waste Disposal Authorities (WDAs), and Waste Collection Authorities (WCAs) became more accountable on the amounts of waste collected and presented for disposal. Local authority waste disposal companies were formed, being an arm's length company of the WDA. Long-hall costs were established to identify the real cost of landfilling one tonne of domestic waste, and the new system gave WCAs a recycling credit equal to this cost. This income now helps the local authority provide recycling systems. Local authorities were given a recycling and/or composting target of 25% by 2000, and the UK's Waste Strategy reaffirms the target to be reached by 2005.

Composting offers one of the greatest opportunities of reaching the target, and under the waste hierarchy is above other disposal options. Home composting is of particular value to WCAs as no cost is involved regarding collection. A free or subsidised home compost unit and a data collection system run by the WCA can easily establish cost reductions in the services offered, but may not count towards any recycling targets.

The collection of garden or green waste is a service for which a charge can be levied by the WCA. The cost for the service is normally based on a sack price. In the Southwest of England where the high number of growing days can prove to be expensive in purchasing sacks for disposal of garden waste, the opportunity of home composting such materials has been met with great enthusiasm.

A renewed interest in gardening and the possibility of producing home made compost has seen many thousands of home composting units put into service; however, recipients are given no written Health and Safety advice and it is fair to say that a number of vulnerable groups may be at risk from the production of composting bioaerosols. These would include individuals suffering from asthma, impaired immune response, patients with leukaemia, Hodgkin's disease or those with drug (corticosteroids, immunosuppressives) or radiation induced immune suppression. Immune compromised patients (TIGR, 2002), Allergic Broncho-Pulmonary *Aspergillosis* (NCBI, 2001), immunocompromised patients (BCCM, 2001), those suffering angioinvasive fungal infections *(Aspergillus Fumigatus, 2002)* and those with allergic diseases, asthma and eczema (SIAF, 2000) are all at risk.

The most frequently encountered opportunistic pathogen is *Aspergillus* (Enviroanalysis, 2000). Emerging evidence from the USA suggests that the vulnerable groups suffering HIV, chemotherapy, and corticosteroids are also at risk from *Aspergillus Fumigatus* (Aerothechlabs, 2002).

In view of this uncertainty, the present investigation has been initiated in order to provide the necessary data for formulating local authority guidance (Lake, 2002).

Methodology

A number of experiments were set up to test the production of *Aspergillus Fumigatus.* The investigation required the mirroring of the home composter as the receptor. A number of different compostable materials were placed into five compost units (Rotol) and an air sampling system was established. Cultures were grown, and colliform forming units (CFUs) identified under the microscope.

The composting units used were identical to those given by local authorities in Cornwall to participating households. It is in the shape of an inverted cone, and as the compost shrinks it falls away from the walls of the unit, and aids in pulling the unit clear of the composting materials for turning. The unit has no bottom and is placed on top of the soil. A tight fitting lid prevents flies and any other visitors from entry and keeps the compost warm. The Rotol is black which aids the exchange of solar energy. Access to the units and prevailing wind direction was assessed and the composters were sited accordingly; one of the units was placed on paving slabs restricting any contact with the soil.

The five contained a mix of materials

- A1 – grass only from lawn mowing
- B2 – grass, hedge and shrub cuttings
- C3 – as in B2, with addition of putrescible waste from the kitchen
- D4 – shrub and hedge cuttings only
- E5 – grass and food scraps.

An air pumping system was devised using a vacuum pump connected to an inline regulator. A flow gauge was inserted between the regulator and the bell jar on the exhaust side. A 12mm0 hose was secured into the lid of the compost unit and fed into the bell jar, and the incoming gases directed onto an agar cover plate. The flow rate was kept constant at 12l-15l per minute to replicate the amount that a human receptor would possibly inhale and respire. The pumping time was limited to one minute, and a settling time of one minute was allowed within the bell jar for any spores to fall out of suspension down onto the agar plate.

Sampling

The agar plates were removed from the bell jar and placed on a bench, and covered to prevent contamination. The temperatures in which the samples were stored were not artificially maintained or controlled but left purely to ruling ambient conditions. Maximum and minimum temperatures were recorded. As the CFUs developed they were counted and when appropriate identified under a microscope. The results were compared with images produced by established authorities. As temperature increased, the aggressiveness of a creamy clear bacteria tended at times to dominate the surface of the sample plate, obscuring other CFUs. The bacteria also attract the fruit fly *Drosophila* (Brody, 2002).

Results

The results of this study show a positive relationship with reference to pH levels, and temperature, confirming the presence of *Aspergillus Fumigatus.* The evidence shows that typical composting pH patterns developed and were associated with an aerobic environment. An initial rise from neutral pH during the mesophilic stage and fall to an acid state during the thermophilic period was followed by a return to neutral pH during the curing stage. pH levels ranged from pH 5.5 – pH 8.3. At the start of composting pH levels ranged from pH 7.3 – pH 8.2 falling to a range of pH 5.5 – pH 6.2 (all Rotols) by the end. Composting temperatures during the field experiments ranged from 8°C – 48.5°C over the five units, including surface and core temperatures. Ambient temperatures ranged from 12°C – 22.2°C and appeared not to influence surface or core temperatures of the compost.

The data indicate an optimum growth and sporulation temperature for *Aspergillus Fumigatus* of between 20°C – 25°C (Fig. 1).

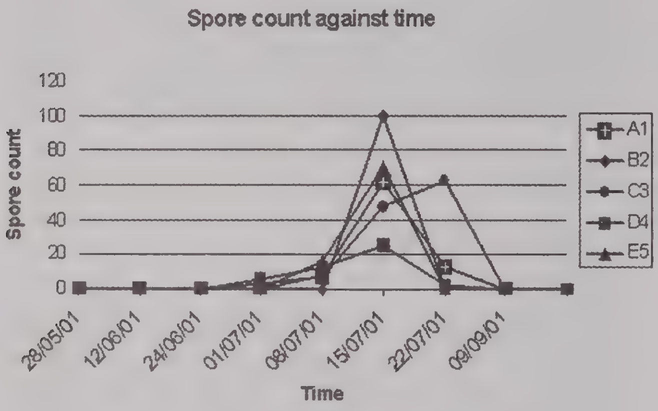


Figure 1. Spore counts from Rotols against time. A1 (grass only), B2 (grass, hedge and shrub cuttings), C3 (as B2, plus putrescible kitchen waste), D4 (shrub and hedge cuttings), E5 (grass and food scraps) all give an *Aspergillus Fumigatus* 'event' during July 2001.

It is noted that the production cycle commences on 24 June 2001, with units A1, B2, D4 and E5 peaking on 15 July, with unit C3 peaking on 22 July. All units became inactive by 9 September 2001. The data also show that B2 filled with grass, hedge and shrub cuttings produced the highest spore count. E5 containing grass and food scraps followed closely, and A1, which also contained a grass element. D4 contained a small amount of grass and produced the lowest spore count. C3 starting from 1 July produced the lowest temperature range, but had the addition of putrescible waste added to the mix. The average pH value remained constant at pH 6.53 for C3 whilst B2 recorded an average pH of 7.18.

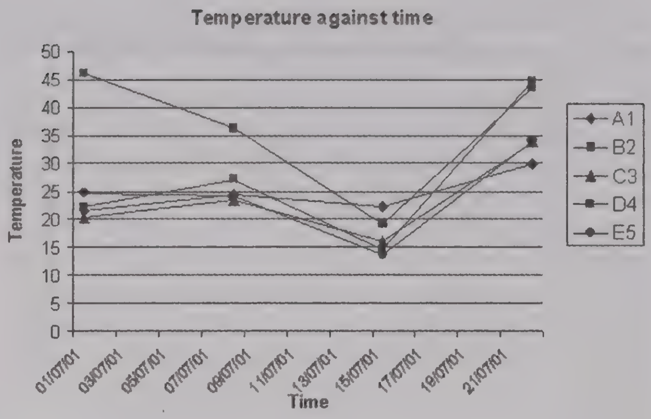


Figure 2. Temperature plotted against time for Rotols, showing dip during July 2001, coinciding with *Aspergillus Fumigatus* peak (Fig. 1).

Figure 2 shows that the temperatures progressively dropped to a low average point of 17°C on 15 July at the time when the spore count peaked. This would suggest that the production of *A/Fumigatus* occurs when the temperatures are much higher. On 1 July 2001 data record the highest temperatures for all units where B2 reached 46°C (Fig. 2). Rotol B2 also records the highest spore count. B2 is filled with grass, hedge and shrub cuttings, and E5 is filled with grass and food scraps. The resulting beneficial nitrogen to carbon ratio in B2 appears to offer the best host environment for the production of *A/Fumigatus.*

Health Effects

Aspergillus Fumigatus colonises widely throughout the world. Its optimum temperature rage for fruiting varies and can have a range from 15°C – 27°C. The spore production is affected if the lower temperatures of the range are experienced. Substrate materials are important for its development as the right mixes of nitrogen, carbon and air determine the temperatures reached within the core areas of the pile.

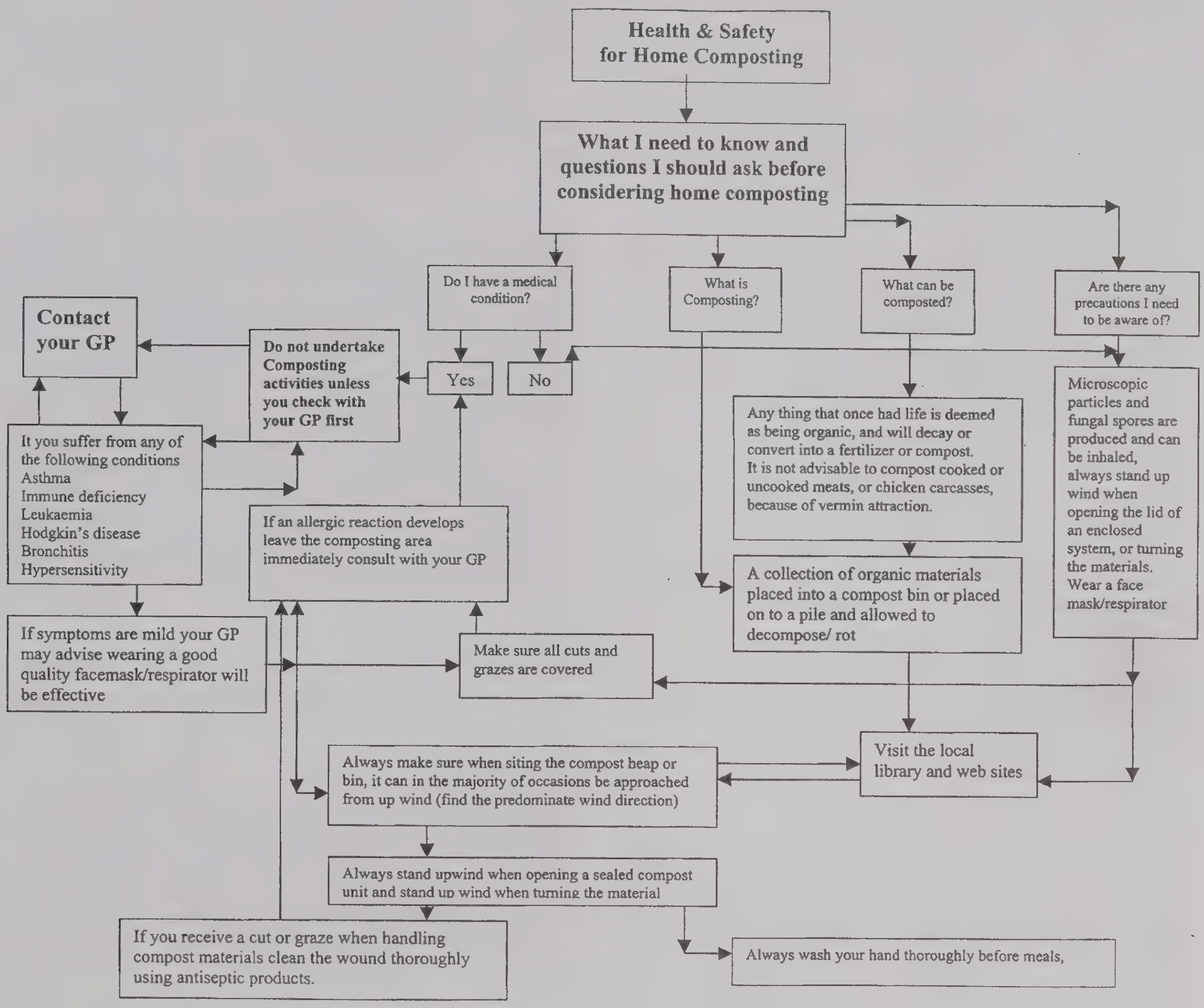
The public and in particular vulnerable groups may be at risk. The closed home composting system does not allow any natural migration of spores to drift away reducing concentrated build-up. Lifting the lid to add materials for composting can release a cloud of spores. The buffeting of new materials against decomposing materials can also create a cloud of fungal particulates.

The spore size of *A/Fumigatus* is approximately 2μm – 3μm and therefore if taken down to the Alveoli or the Bronchiole areas of the respiratory system can cause *Aspergillosis.* Environmental factors are known to trigger asthma attacks. Pollens, house dusts and ambient air pollution are also accepted among other triggers, and hay fever is a very common culprit. This links back to the agricultural workers and *Aspergillosis.* Although air pollution has not been established as a cause, a health and safety flow chart is proposed that would be given to all members of the public on purchasing a home composting unit (See Flow Chart 1). It provides step-by-step guidance for composting schemes and might prove a starting point in other aspects of bioaerosol risk management.

Conclusions

Aspergillus Fumigatus has been detected in all the types of home composting used in Cornwall. It has been readily

Proposed Flow-Chart guidance intended for members of the public considering home composting and any associated health risks


Health & Safety for Home Composting
What I need to know and questions I should ask before considering home composting
Do I have a medical condition?
What is Composting?
What can be composted?
Are there any precautions I need to be aware of?
Contact your GP
Do not undertake Composting activities unless you check with your GP first
Yes
No
It you suffer from any of the following conditions Asthma Immune deficiency Leukaemia Hodgkin's disease Bronchitis Hypersensitivity
If an allergic reaction develops leave the composting area immediately consult with your GP
Any thing that once had life is deemed as being organic, and will decay or convert into a fertilizer or compost. It is not advisable to compost cooked or uncooked meats, or chicken carcasses, because of vermin attraction.
A collection of organic materials placed into a compost bin or placed on to a pile and allowed to decompose/ rot
Microscopic particles and fungal spores are produced and can be inhaled, always stand up wind when opening the lid of an enclosed system, or turning the materials. Wear a face mask/respirator
If symptoms are mild your GP may advise wearing a good quality facemask/respirator will be effective
Make sure all cuts and grazes are covered
Visit the local library and web sites
Always make sure when siting the compost heap or bin, it can in the majority of occasions be approached from up wind (find the predominate wind direction)
Always stand upwind when opening a sealed compost unit and stand up wind when turning the material
If you receive a cut or graze when handling compost materials clean the wound thoroughly using antiseptic products.
Always wash your hand thoroughly before meals,

collected in the free air space of Rotols during the summer months where and when fugitive emissions are likely to occur. Although levels appear relatively low, a flow-chart approach is now proposed as guidance for the most vulnerable groups.

Acknowledgement

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Martin Lake, Carrick District Council, Pydar Street, Truro TR1 1EB.
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Swedish Experience with Low Emission Zones

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In 1996 Stockholm introduced an "environmental zone" in the city centre. The main purpose was to encourage the use of cleaner vehicles, improve air quality and to some degree reduce noise. The zones targeted trucks and buses.

This paper describes the effect of the environmental zone on the emissions and atmospheric levels of nitrogen oxides and particulate matter. According to these calculations heavy-duty vehicle emissions within the zone of NO_x and particulate matter have been reduced by about 10% and 40% as compared to the situation without the zone. The corresponding reductions for atmospheric concentrations are up to 2% for NO_x and up to 9% for $PM_{0.2}$.

Introduction

In Sweden, the municipalities have formal responsibility for air quality management in the cities. In Stockholm road traffic is the main source of most air pollutants, and significantly harms both public health and the environment. There is no one single action that would solve the air pollution problems related to road traffic. The City seeks, by a variety of means, to limit harmful effects of traffic pollution. In 1996, Stockholm, Gothenburg and Malmö, the three largest cities in Sweden, introduced environmental zones in their city centres. The main purpose was to speed up the transition to cleaner traffic, improve air quality and, to some degree. reduce noise.

In this paper we present an assessment of the effects on emissions and air quality of the environmental zone in Stockholm. An air quality management system with a detailed emission database and an air quality dispersion model is used to compare a situation without the zone, with the current situation year 2000 with the zone. A scenario with no illegal heavy-duty vehicles in the zone is also included.

Implementation of Environmental Zones in Sweden

Vehicle regulation

All trucks and buses (>3.5 tonnes) travelling in the zones must meet the Euro I emission standard (EU Directive 91/542/EEC Level A). Other vehicles that may travel in the zone are

- Vehicles from 9 and 15 years old that have been retrofitted with a certified emission control device. Approved control devices are catalytic converters in combination with particulate traps.
- Old vehicles where the motor has been replaced with a new.
- Vehicles that travel only very seldom within the zone. Such vehicles need a special permit from the Road and Traffic Administration of the City.

The Swedish Motor Vehicle Inspection Company is responsible for the approval of vehicles with or without retrofitted emission reduction kits. The zone in Stockholm has been in force for 9 years and a new regulation is now being implemented from 2002 onwards. The effectiveness of the zone in Stockholm is evaluated in a special control programme which includes visual inspections four times per year at two sites in the city centre. Each time around 100 trucks and buses are registered. Based on the visual inspections a database has been created which includes information on the age, weight, vehicle length, emission control system and model of each vehicle. During 2000, 470 vehicles were registered in the database. All trucks and buses must have a sticker placed on the windshield. Illegal driving in the zone may result in a fine enforced by police authorities.

The geographic area

Figure 1 shows the extent of the environmental zone area of Stockholm. Note that some of the major roads within the Environmental zone area have been exempted from the regulations (Valhallavägen, Lidingövägen, Klarastrandsleden etc.). The area covers the most densely populated area in the city centre with about 220 000 people, which is approximately 30% of the total population in the city of Stockholm.

Methodology

Calculated emissions

The calculations are based on a detailed emission database administered through a Regional Association for Air Quality Management of the counties of Stockholm and Uppsala (www.slb.mf.stockholm.se/lvf). The database includes 4 500 road-links for which traffic volume, speed, vehicle composition, driving conditions etc. are defined. There are 45 road types, each containing information on the temporal distribution of 5 vehicle types[1, 2]. The estimates of total traffic volumes are primarily based on measurements in situ. Such measurements are of different kinds: regular automatic traffic counting by local traffic and street authorities within municipalities, automatic traffic counting on main roads by the Swedish National Road Administration and manual surveys of traffic volumes. Traffic volumes on major roads are taken from information provided by the Swedish National Road Administration. For other roads information in the database relies on the traffic volumes estimated from the measurements administered by the local municipalities. Heavy-duty vehicle (HDV) emissions in the zone are based on the control program described above.

Emission factors are mainly based on the "EVA model" (version 2.2) of the Swedish Road and Traffic Administration. Emission factors for ethanol buses are based on [3]. There are five vehicle types within the zone:

- Light-duty gasoline cars
- Light-duty diesel cars
- Medium-duty gasoline trucks (‹3.5 ton)
- Heavy-duty diesel trucks and buses (›3.5 ton)
- Ethanol buses

For each vehicle type there are six different categories corresponding to different emission standards. The additional emissions due to cold starts in the model are large for small roads (a journey of about 10 km for each cold start) and almost zero for motorways (a journey of about 200 km for each cold start). Cold start emissions are implemented as line sources distributed along different road types.

Table 1 shows that 1.6 ton/year, or 12%, of the total vehicle exhaust emission of particulate matter was due to exhaust emissions from heavy-duty vehicles (trucks and buses) in the environmental zone 2000. The single most important vehicle category is gasoline cars with 53% of the total exhaust emission of particulate matter in the zone. For NO_x 32% (170 ton/year) of the total exhaust emission was due to heavy-duty vehicles, and 62% (330 ton/year) was due to light-duty exhaust emissions.

Most of the exhaust particles from diesel and gasoline vehicles have diameters less than 2.5 μm (70% to 95% of the total mass) and almost all are less than 10 μm[4]. Re-suspension of road dust and other particle sources such as particle emissions due to wear of brake linings have not been included in these emissions, since they are not affected by the environmental zone regulations. (But re-suspension is included in the dispersion calculations, see below).

The emission database includes information on point and area sources in the two counties[2]. It is updated yearly through a co-operation between the municipalities and two regional bodies that are members of the Association.

Dispersion model

A Gaussian dispersion model is used to calculate annual concentrations. Dispersion calculations are based on a wind field obtained from a diagnostic wind model. The horizontal large-scale wind is assumed to be uniform in the model. The area-wide wind field is estimated assuming geostrophic balance. The model is not mass conservative and vertical circulation cannot be described, since it is a one-layer model. The model cannot describe the sea breeze. But it can diagnose it, if mast data detect it. The model does not resolve individual buildings. Instead, surface structures like houses and trees enter the model through local roughness values and through the wind field. In an open area the calculation height is 2 m above ground level. Over a city the values will reflect the concentrations at 2 meters above roof height. A special treatment of the Gauss model plume length is introduced to avoid unrealistically long plumes. This length depends on the stability and persistency of weather conditions. A detailed description of the model is given in the Airviro User Documentation[5]. A climatology with 360 weather cases has been created based on 10 years of meteorological measurements in a 50 meter mast in the southern part of the city. The average concentration of these 360 cases represents the annual average.

Model calculations of NO_2

The Gaussian dispersion model does not consider the photochemistry of NO, NO_2 and O_3. Instead an empirical relationship has been utilised to convert calculated concentrations NO_x to NO_2 concentrations. This procedure is used for long term average values and has been validated by comparing with measured NO_2 concentrations at a number of sites in the county of Stockholm[1].

Table 1. Total emissions in the environmental zone of Stockholm and average emission factors for road traffic in the zone area. The emission factors are averages for the traffic composition and driving conditions during year 2000 with zone regulations. For particulate matter only exhaust emissions are included. For NO_x values are given as NO_2.

	Particulate matter		***NO_x***	
Vehicle type	***g per vehicle km***	***Ton/year***	***g per vehicle km***	***Ton/year***
Light duty gasoline car	0.018	7.0 (53%)	0.85	330 (62%)
Light duty diesel car	0.15	3.3 (25%)	0.74	16 (3.0%)
Medium-duty gasoline trucks (‹3.5 ton)	0.056	1.2 (9.4%)	0.85	18 (3.4%)
Heavy-duty diesel trucks and buses (›3.5 ton)	0.071	1.1 (8.1%)	5.7	86 (16%)
Ethanol buses	0.054	0.5 (4.0%)	8.4	84 (16%)
Sum		**~13**		**~540**

Model calculations of particulate matter (PM_{10} and $PM_{0.2}$)

The total PM_{10} annual concentration was calculated as the contribution from all local sources plus the concentration in background air as measured at a rural site 75 km south of Stockholm. Based on model calculations and measurements it has been found that the contribution from re-suspension of road dust to PM_{10} was about equal to that from the total exhaust emission of road traffic[1]. Under certain conditions nucleation may be a source of new particles but their mass is sufficiently small that they do not significantly alter the total aerosol mass concentration on the time-scale for dispersion of air pollutants in urban areas[6, 7]. Condensation/evaporation of inorganic and organic compounds increases or decreases the particle mass and changes the size of the particles. Two main sources of condensation/evaporation are sulphur dioxide (SO_2) and nitric acid (HNO_3). Emitted SO_2 is oxidised to sulphate, which condense onto droplets and particles. Most particulate sulphate is found as a combination of sulphuric acid, ammonium bisulphite (NH_4SO_4), and ammonium sulphate ($(NH_4)_2SO_4$). The time-scale for gas-phase transformation of SO_2 to sulphate range from less than 0.01% (of gaseous SO_2) per hour to 5% per hour, depending on meteorological and photochemical conditions. Due to low emissions and concentrations of SO_2 and the short residence time of the air in the urban area of Stockholm the contribution of locally emitted SO_2 is assumed to be negligible for PM_{10} concentrations in this region.

Nitric acid, formed as a result of the oxidation of NO_2, may be absorbed on droplets and particles. Particulate nitrate (in the form of ammonium nitrate) may evaporate from the particles to the air, due to equilibrium between ammonium nitrate and gaseous nitric acid and ammonia. Nitric acid may also react with sodium chloride and alkaline dust particles forming other particulate nitrate species. The rate of gas-phase conversion of NO_2 to nitric acid ranges from less than 1% per hour to 90% per hour. Emitted NO_2 may thus, under favourable conditions, be converted to particulate nitrate and thereby contribute to the local PM_{10} levels in Stockholm. This is not considered in the model calculations. Measurements show that the levels of nitric acid and particulate nitrate are around 0.5 $\mu g/m^3$ higher in central Stockholm compared to rural (background) areas. Whether this is due to local formation of nitrate from NO_2 or primary emission is not known, but it gives an upper limit to the effect of local NO_2 to nitrate conversion.

There is also a contribution to the PM from organic compounds, either primary emitted or formed in photochemical processes (secondary organic compounds). Recent studies by [6] and [7] show that the contribution from secondary organic material to the fine particle mass in San Francisco is small compared to direct emission of organic material. Since it may safely be assumed that photochemical processes are less important in Stockholm compared to San Francisco, photochemical formation of secondary organic compounds should be even smaller in Stockholm. Primary semi-volatile organic compounds may also be adsorbed or absorbed on the aerosol particles, but since there is very limited information on the levels of these compounds in Stockholm this is not considered in our calculations. Coagulation processes affect the size-distribution of the aerosol. However, since wet and dry deposition of particles is not considered here, the size distribution is not taken into account. When haze or fog exists, particles grow to larger sizes[8]. Particle growth and shrinkage are determined by the amount of water transferred to and from the aerosol. These processes depend on the relative humidity, temperature and chemical composition of the aerosol. Deposition of fog droplets may be large. In addition, fog reduces the photochemical reactions and increases the reaction rates above the fog layers[7]. The importance of such processes for particle concentrations in Stockholm has not been assessed.

The total annual concentration of PM_{10} is converted to $PM_{0.2}$ based on measurements of the particle size distribution at a roof level site during March 2000. It was assumed that the fraction $PM_{0.2}$ was the same in the whole city and representative for the annual average.

Emission scenarios

Three different scenarios have been considered:

- Present situation (year 2000) with environmental zone
- Year 2000 without the zone
- Year 2000 with the zone but no illegal vehicles (100% of the heavy-duty vehicles drive according to zone regulations)

Results

Emissions year 2000

The total heavy-duty traffic volume in the zone was 25 million vehicle kilometres (Mvkm). This is about 5% of the total traffic volume (500 Mvkm). The average age of these vehicles was 6 years. The inner city was trafficked by 250 ethanol driven buses constituting 11.5 Mvkm/year. The average age of the buses was 4 years. Most the traffic volume was due to new vehicles that also had the lowest emission factors.

During 2000 visual inspections of 470 vehicles showed that about 90% of the heavy-duty trucks were driving in accordance with the regulations for the zone (Figure 2). Most of these trucks and buses (80% of all vehicles) were 1992-2000 year models. Ten percent were older vehicles which had installed certified exhaust control device or vehicles with a new motor. The buses were mainly ethanol driven buses, part of the local network of the city of Stockholm. Only five percent of the heavy-duty vehicles were diesel buses (transfer- and tourist buses).

Particulate matter

Figure 3 shows the emissions of exhaust particulate matter from different groups of heavy-duty vehicles with the zone

2000. To be able to compare the emissions with and without the zone, the same fractions for the respective groups (e.g. "illegal vehicles", "exhaust after treatment") was applied on the vehicle fleet without the zone. The distribution between the groups was obtained from the manual inspection programme in 2000.

Emissions of exhaust particles from heavy-duty vehicles in the zone 2000 were about 40% lower as compared to a situation without the zone (Figure 3). The most important effect is due to the decrease in the number of old trucks in the vehicle fleet leading to about 30% lower emissions of the approved vehicle group ("Approved 2000 – 1992"). For the trucks which had installed exhaust after treatment the reduction in emissions was 87%, but since their total emissions are smaller the contribution to the total emission reduction is only about 10%. The manual inspection program has shown that 10% of the trucks drive illegally in the zone. These old trucks are estimated to give rise to about 25% higher total PM emissions as compared to a situation with no illegal driving.

It has been estimated that about one third (3.4 Million vehicle km) of the total traffic volume of the inner city buses would have been due to diesel buses without the zone. The zone regulations have led to their replacement with ethanol buses, resulting in a 50% reduction of emissions from all inner city buses.

NO$_x$ emissions

For NO$_x$ the effect on heavy-duty vehicle emissions is not as large as for particulate matter. Most of the reduction in emissions is due to the decrease in the number of old trucks in the vehicle fleet leading to about 25% lower emissions of the approved vehicle group ("Approved 2000 – 1992"). There is also a small reduction due to the replacement of motors in old trucks. Installation of exhaust after-treatment equipment on old vehicles does not have any effect on NO$_x$ emissions. The emission factors for new ethanol buses are 30% lower than the emission factor for diesel buses.

The overall effect is that NO$_x$ emissions from heavy-duty vehicles are 10% lower in the zone as compared to a situation without the zone.

Concentrations due to all road traffic in the zone

Firstly, calculations were made to see the total contribution to ambient concentrations of particulate matter and NO$_x$ from **all** vehicle emissions within the zone. For NO$_x$ the contribution of the total road traffic in the environmental zone varies from 1% of the total concentration in the suburban areas to 76% of the total concentration in the centre of the city (see Table 2 and Figure 5). The population weighted* average concentration due to road traffic within the zone is 15 $\mu g/m^3$. This corresponds to 36% of the population weighted total concentration of the inner city of Stockholm.

Exhaust emissions of NO$_x$ due to heavy-duty trucks within the zone contribute 2.5 $\mu g/m^3$ (population weighted mean concentration) while light-duty vehicles (cars and light-duty trucks) contribute 10 $\mu g/m^3$. These values correspond to 17% and 67% of the total concentration within the zone, respectively. Exhaust emissions from ethanol buses contribute 2.6 $\mu g/m^3$.

For particulate matter exhaust emissions due to all road traffic in the zone contribute with a population weighted average concentration of 0.39 $\mu g/m^3$. The population weighted $PM_{0.2}$ concentration was 1.1 $\mu g/m^3$. The average contribution from exhaust emissions due to road traffic in the environmental zone was 36% of the total $PM_{0.2}$ concentration (Table 2). The geographic variation of the contribution from all road traffic and only heavy-duty vehicles are presented in Figure 6.

Exhaust emissions from trucks give a population weighted concentration of 0.041 $\mu g/m^3$, corresponding to 3.8% of the total $PM_{0.2}$ concentration in this area. This may be compared with the contribution due to exhaust emissions of particulate matter from light-duty vehicles of 0.32 $\mu g/m^3$ (29% of the total $PM_{0.2}$ concentration).

Table 2. Calculated contributions to the concentrations of NO$_x$ and particulate matter from exhaust emissions due to road traffic within the environmental zone 2000. The values represent roof level and the resolution is 100 x 100 metres. The upper value is a population weighted average and the values within brackets correspond to the minimum and maximum values inside the model domain.

	Absolute contributions in $\mu g/m^3$			Percentage contribution to the total concentration without environmental zone.		
	All road traffic	Only heavy duty vehicles	Only cars and light duty trucks	All road traffic	Only heavy duty vehicles	Only cars and light duty trucks
NO$_x$	**15.3** [0.4 – 61]	**2.5** [0.1 – 10]	**10.2** [0.2 – 41]	**36 %** [1% – 76%]	**7%** [0.2% – 13%]	**26%** [0.5% – 48%]
$PM_{0.2}$	**0.39**[2] [0.01 – 1.6]	**0.041**[2] [0.001 – 0.16]	**0.32**[2] [0.008 – 1.3]	**35%**[1] [1% – 100%[3])]	**3.8%**[1] [0.1% – 13%]	**29%**[1] [0.8% – 100%]

[1] Percent of total $PM_{0.2}$ concentration.. $PM_{0.2}$ has been estimated based measured particle number distribution assuming spherical particles with a density of 1.5 g/cm^3.

[2] Only exhaust particles are included in the calculations and all exhaust particles are assumed to be less than 0.2 μm in diameter.

[3] In a small area in the city centre the contribution becomes more than 100% due to uncertainties in the calculations (see text).

* Sum of the concentrations in each grid times the total population in the grid divided by the total population.

Table 3. Calculated reductions in the concentrations of NO_x and particulate matter due to the introduction of the environmental zone in Stockholm. The values represent roof level and the resolution is 100 x 100 metres. The upper value in bold is a population weighted average and the values within brackets correspond to the minimum and maximum values inside the model domain.

	Reduction in the current situation 2000 compared to a situation without the zone		Reduction in a situation with 100% legal vehicles 2000 as compared to a situation without the zone	
	µg/m³	%	µg/m³	%
NO_x	**0.52** [0.07 – 1.8]	**1.3%** [0.6% – 2.2%]	**0.60** [0.1 – 2.5]	**1.5%** [0.7% – 2.5%]
$PM_{0.2}$	**0.032** [0.05 – 0.11]	**3.0%** [0.5% – 9%]	**0.043** [0.05 – 0.15]	**4.0%[1])** [0.5% – 12%]

[1] Calculated as percentage of total $PM_{0.2}$ concentration.

Reduction in concentrations due to the zone

A summary of the calculations of the effects on the concentrations of NO_x and particulate matter is given in Table 3. The table shows how much lower the concentrations are (1) with the zone 2000 and (2) with a fully implemented zone (i.e. no illegal vehicles) as compared to a situation with no environmental zone. All values are annual averages and refer to roof level.

In the situation with the zone 2000 the population weighted NO_x concentration is calculated to be 0.52 µg/m³ lower as compared to a situation without the zone (Table 3). This corresponds to 1.3% of the total concentration with the zone 2000. It is about 20% of the reduction that would be obtained if all exhaust emissions from trucks and buses in the zone were eliminated. The fact that there are 10% heavy-duty vehicles that drive illegally in the zone does not make much difference.

For particulate matter the concentration in the current situation with the zone is calculated to be 0.032 µg/m³ lower compared to the situation without the zone. This corresponds to ca 3% of the total population averaged $PM_{0.2}$-concentration. It is 44% of the maximum possible reduction if all exhaust emissions from heavy-duty vehicles were eliminated. If there were no illegal heavy-duty vehicles driving in the zone, the reduction would be 0.043 µg/m³, as compared to the situation without the zone. This corresponds to ca 61 % of the maximum possible reduction, i e. if all exhaust emissions due to heavy-duty vehicles were eliminated within the zone).

The geographical variation is essentially identical to the one shown in the figures above. The largest effects of the zone are obtained in the centre of the city, which is the most densely populated part of the area and where most people work.

Discussion: Number or mass of particles?

So far, most measurements of particulate matter in urban areas and also in the city of Stockholm refer to the mass concentration (µg/m³). Measurements of the particle size distribution in Stockholm are presented in Figure 7. These data indicate that exhaust emissions due to local road traffic have a large impact on the number concentration of particles less than about 200 nm (diameter) due to exhaust emissions. There is also an important effect from local road traffic on the number of particles larger than about 600 nm (due to road dust re-suspension). But the influence on particle sizes between 200 nm and 600 nm is much smaller.

The number of particles emitted in the exhaust of diesel vehicles is several times higher compared to the exhaust of gasoline vehicles[9]. Based on measurements in a road traffic tunnel in Stockholm it has been estimated that heavy-duty vehicles emit at least 15 times more particles per vehicle kilometre compared to light-duty vehicles[10].

The influence of heavy-duty vehicles on the number of particles in Stockholm may thus be much more important than if the mass concentration is considered. It has been shown that emissions of both the mass and number of exhaust particles from new heavy-duty trucks and buses are lower than those from old vehicles[11]. In addition, the use of a CRT™ filter ("Continuously Regenerating Trap") on old vehicles may remove 99% of the particles of all sizes larger than 30 nm in transient driving cycles. An oxidation catalyst did not have any effect on the number of particles. In Stockholm CRT filters are the most common exhaust after treatment equipment. Thus, the effects on the concentrations of particles by the introduction of the environmental zone may be larger than indicated in the evaluation presented here if the number of particles had been considered. So far there are no data that may confirm this, but the regular environmental monitoring programme in Stockholm now includes measurements of particle number concentrations.

Conclusions and Discussion

The environmental zone in Stockholm has reduced both emissions and ambient concentrations of a number of pollutants. However, even though the effects on total emissions from heavy-duty vehicles may be substantial, the corresponding effect on concentrations may be very small compared to the total concentrations due to all sources. This report describes the effects on NO_x and particulate matter. For NO_x heavy-duty vehicles account for 32% of the total emissions within the zone area. As a result of the implementation of the environmental zone, emissions from heavy-duty vehicles have been reduced by about 10% compared to the emissions without the zone. Even though new heavy-duty vehicles have lower NO_x emissions, exhaust after-treatment has little or no effect on the emissions. The corresponding reduction of total NO_x concentrations is estimated to be 0.5% to 2% of total NO_x levels. Most of the NO_x emission in the zone area comes from light-duty gasoline vehicles (~60%).

For particulate matter, heavy-duty vehicle emissions have been reduced by 40% due to the implementation of the environmental zone. The importance of this for the ambient concentration depends on which fraction of the particulate matter is used as reference. The reduction of the ambient concentrations determined as $PM_{0.2}$ (mass of particles with a diameter less than 0.2 μm) due to the implementation of the environmental zone is estimated to be between 0.5% and 9% in the area. The reduction compared with the total levels of PM_{10} is much less since a substantial part is due to re-suspension of road dust and long-range transport. It seems likely that the environmental zone has led to a significant reduction of the particle number concentration. This is due to three reasons:

- Measurements in a road tunnel in Stockholm and other data show that emissions from old heavy-duty diesel vehicles without an exhaust filter are much larger than light-duty gasoline vehicles.
- New heavy-duty vehicles and old vehicles with filters emit much less particles than old.
- The total particle number concentration close to roads inside the zone is predominantly due to local emissions from road traffic – the number concentration in background rural air is several orders of magnitude lower.

The lack of data on the physical and chemical properties of urban PM have made it difficult to explain correlations between PM and various health effects as observed in epidemiological studies. It is not yet known which property of the urban aerosol is most important from a health point of view, but respiratory health effects are associated with the number of ultra fine particles[12]. The target parameter in air quality regulations in Europe refers to PM_{10}. In Stockholm, and probably many other cities in Northern Europe, exceedances of the daily average limit value (50 μg/m^3, allowed to be exceeded only 35 times per year) is mainly due to road dust resuspension during late winter and spring. Exhaust emissions of particles from local road traffic make a very small contribution to PM_{10} concentrations during these periods.

Finally, it should be pointed out that the effect of the low emission zone on air quality is likely to be large when compared to other actions which it is possible for local city administrations to implement. Most of the potential actions that have been examined lead to much less than a percent reduction of total air concentrations.

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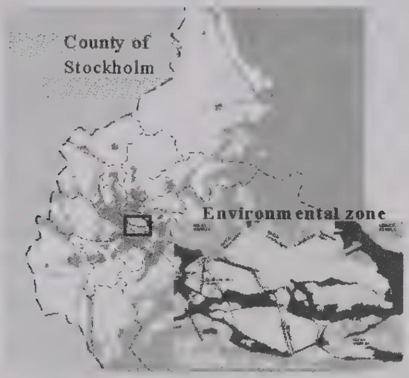

Figure 1. Map showing the Environmental zone area of Stockholm. Major roads have been exempted from the zone regulations.

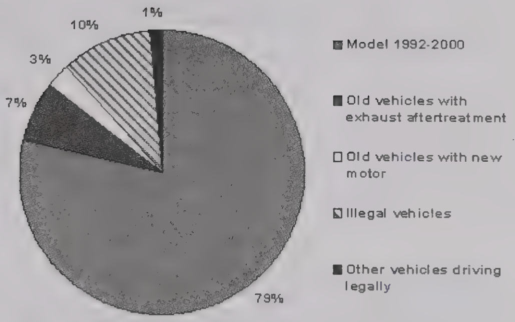

Figure 2. Heavy duty vehicle composition according to manual control of 470 vehicles in the zone in January, April, July and October 2000.

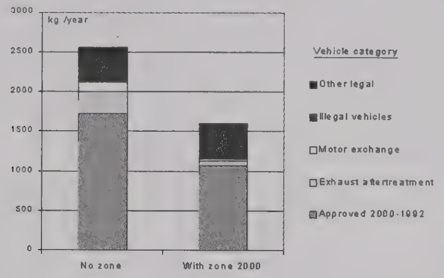

Figure 3. Emission of particles with and without the zone 2000 from heavy-duty vehicle categories inside the zone area. To be able to compare the emissions with and without the zone, the same fractions as determined with the zone, for the respective groups (e.g. "illegal vehicles", "exhaust after treatment") was applied on the vehicle fleet without the zone.

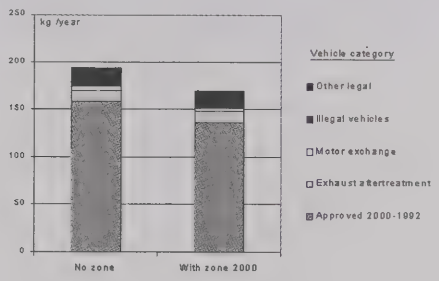

Figure 4. Emission of nitrogen oxides (ton NO_2/year) from heavy-duty vehicles inside the zone for three different scenarios. To be able to compare the emissions with and without the zone, the same fractions as determined with the zone, for the respective groups (e.g. "illegal vehicles", "exhaust after treatment") was applied on the vehicle fleet without the zone.

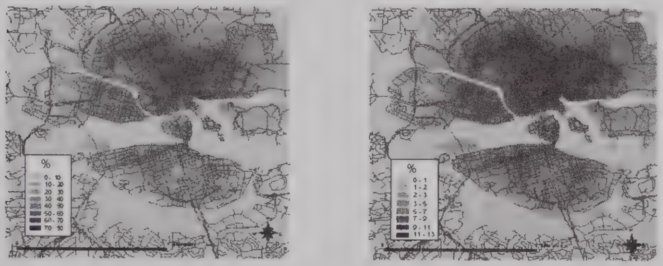

Figure 5. Geographic variation of the contribution to NO_x levels from exhaust emissions due to all road traffic within the environmental zone 2000 (left panel) and only trucks and buses (right panel). The values represent roof level and are given as percent of the total concentration.

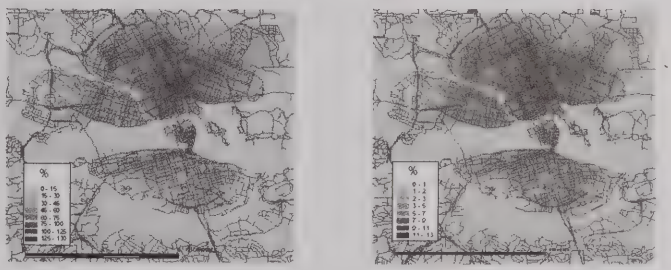

Figure 6. Geographic variation of the contribution to $PM_{0.2}$ levels from exhaust emissions due to all road traffic within the environmental zone 2000 (left panel) and only trucks and buses (right panel). The values represent roof level and are given as percent of the total concentration of $PM_{0.2}$.

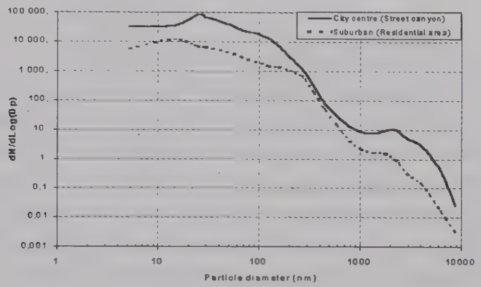

Figure 7. Particle size distribution of the number of particles at 2 sites in Stockholm during March 2000. Solid line: city centre (street canyon, 40 000 vehicles per day), dashed line: suburban site (residential area, 1 000 vehicles per day).

Economic Costs of Health Effects of Vehicle Emissions

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The objective of this paper is to outline the state of the art with regards to the economic valuation of the health effects of vehicle emissions. The paper therefore briefly sets out the rationale for this economic valuation before describing the most recent estimates that have been made of these economic costs in the UK and current progress in the key methodological issues involved in the valuation exercise.

Introduction

Why do we need to assess the economic costs of the health effects of air pollution?

Vehicle emissions have long been recognised as contributing to health related air quality effects and these health effects – illness and premature death – have been documented in numerous studies[1]. Those pollutants which give the greatest cause for concern are carbon monoxide (CO), hydrocarbons (HC), sulphur oxides (SO_x), nitrogen oxides (NO_x), particulate matter (PM), lead (Pb), and secondary pollutants, particularly ozone (O_3). Inhalable suspended PM, for example, particularly particles less than 10 microns in size (PM_{10}), can pass through the natural protective mechanism of the human respiratory system. The smallest particulates (2 microns or less – $PM_{2.5}$) are created primarily by the combustion of petroleum fuels. The World Bank (1992) for example, estimates that reducing suspended PM to safe levels could reduce premature deaths by 300,000 to 700,000 a year in developing countries alone. It is also estimated that 500,000 new cases of chronic bronchitis and many lesser health effects each year result from anthropogenic-produced particulates.

As a consequence of these health effects, there is a recognition that policy actions at local, national and international levels should be informed by the assessment of the associated economic effects. These assessments generally take the form of cost-benefit analyses of the implementation of policy measures. Draft EU Air Quality Directives that involve the imposition of air quality standards for individual pollutants are legally required to be subject to cost-benefit analysis. The monetary evaluation of these health effects is an essential input to the policy formulation process since they comprise a major component of the benefits (avoided costs) that may result from such policies. Whilst it is not a statutory requirement for cost-benefit analysis of proposed environmental legislation in the UK, it is well understood that the assessment of economic benefits is increasingly being made in monetary terms.

What are the principal health effects that we wish to value?

The suspected health effects of ambient air pollution are summarised in Table 1; a distinction is made between those which are presently quantifiable and those which are not. This indicates the scope for health damage assessment. As one can see from Table 1, the full range of suspected impacts cannot currently be quantified.

Table 1. Suspected Health Effects of Ambient Air Pollution

Quantifiable	Not-quantifiable
Mortality – acute and chronic	Neonatal mortality
Bronchitis – acute and chronic	Non-bronchitis chronic respiratory disease
New asthma cases	Behavioural effects (e.g. learning disabilities)
Respiratory Hospital Admissions (RHA)	Neurological disorders
Cardiovascular Hospital Admissions (CHA)	Allergies
Accident & Emergency Visits for asthma	
Lower respiratory illness	
Upper respiratory illness	
Shortness of breath	
Respiratory Symptoms	
Restricted Activity Days	
Days of work loss	
Asthma status	

Source: US EPA (1999) in Davis, Krupnick and Thurston (1999)

Quantified Assessment of the Health Costs of Air Pollution

In current quantified assessments of the health costs of air pollution, a dispersion model allows air pollution concentrations in an area to be calculated as a function of time and location. Following this, an exposure and damage assessment is conducted to provide information on the resulting impact of air pollution on human health (and damage to ecosystems, buildings and materials). The exposure, or impact, is defined as the product of the local air pollution concentration, within a grid square for example, and the number of susceptible objects, such as the population, within that location. The assessment of health damage in physical terms therefore requires knowledge of dose-effect (more commonly known in the literature as dose-response) relationships and exposure distributions for populations.

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[1] See, for example, Working Group on Public Health and Fossil Fuel Consumption (1997); Cropper *et al* (1997); Ostro (1994); Ostro *et al* (1995 and 1996); Schwartz and Dockery (1992); Dockery and Pope (1993); and Pope *et al* (1995).

Finally, an economic assessment may be undertaken to value changes in the damaging effects of air pollution, and the costs of environmental protection avoided. The monetary assessment of damages is achieved using various methods such as the valuation of productivity losses, defensive or averting expenditures, and willingness-to-pay (WTP) and/or willingness-to-accept (WTA) compensation for changes in environmental quality.

The most sophisticated approaches to implement this framework rely on the so-called 'impact-pathway' approach. An impact-pathway describes the sequence of events linking the emission of a specific pollutant to a specific impact and subsequent valuation. Aggregate costs/benefits associated with a change in the source mix are found by summing over all affected pollutants and impacts of interest. Figure 1 presents a framework for air quality assessment.

In general, with the impact-pathway approach, valuing the cost of pollution-induced impacts (expressed in terms of health effects) is a two-step process[2]:

1. Dose-response functions, which relate the level of air pollution to the incidence of particular health outcomes (types of health impact), must be established, from which changes in the number of additional cases of that health outcome can be estimated.
2. The cost of the pollution-induced health outcome can then be estimated by multiplying the estimated change in the number of cases by the corresponding economic (unit) value.

Economic Assessment

Valuation of Damages: Impact-pathway Approach

The starting point for the valuation of health end-points is the identification of the components that comprise changes in welfare. These components should be summed to give the total welfare change, assuming no overlap between categories. The three components include:

(i) *Resource costs* i.e. medical costs paid by the health service in a given country or covered by insurance, and any other personal out-of-pocket expenses made by the individual (or family).

(ii) *Opportunity costs* i.e. the cost in terms of lost productivity (work time loss (or performing at less than full capacity)) and the opportunity cost of leisure (leisure time loss) including non-paid work.

(iii) *Dis-utility* i.e. other social and economic costs including any restrictions on or reduced enjoyment of desired leisure activities, discomfort or inconvenience (pain or suffering), anxiety about the future, and concern and inconvenience to family members and others.

There are two broad ways of estimating WTP/WTA-based measures of morbidity costs. It can be inferred either from an individual's ***averting behaviour,*** or from an explicit expression of what the individual is WTP/WTA (as compensation) to avoid (accept) the illness, in which case we are talking about ***contingent valuation*** (CV) studies[3]. As with morbidity outcomes two approaches are generally used to derive WTP/WTA-based values for premature mortality: ***hedonic wage-risk*** studies[4] and ***contingent valuation*** studies[5]. Measures of the value of a reduction in a health risk valued in these ways are preferable from an economists perspective. In the absence of such studies, more 'direct' measures are commonly used to value morbidity and mortality – these are the ***cost of illness*** approach[6] for morbidity, and the ***human capital*** approach[7] for *mortality.*

Table 6 below provides a sample of the central, as well as low and high, economic unit values typically used in the valuation of health outcomes. (All values have been expressed in US$1990). These unit values represent those used in four major studies conducted in the US, Canada and Europe. For convenience[8], individual WTP for mortality risk and chronic risk reductions are expressed in terms of the value of a statistical life (VSL) and the value of a statistical case of chronic disease (VSC), respectively. The relatively lower VSL for TAF and AQVM reflect the fact that these values have been adjusted for 'age effects', discussed below.

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[2] This two-step process is also applicable to the valuation of damage to agricultural crops, buildings and materials.

[3] The latter was used by Alberini et al (1995) in Taiwan. CSERGE et al (1999) also used contingent valuation to derive morbidity cost estimates in five European countries.

[4] **Hedonic** analysis assumes that, other things being equal, workers in riskier jobs must be compensated with higher wages than those in safer jobs. For example, a wage premium of US$ 100 may be necessary to get workers to undertake a job with a 1-in-10,000 chance of increased mortality. This yields an implicit 'value of a statistical life' (VSL) of US$ 1 million.

[5] **Contingent valuation** involves asking people directly, for example, what they would be willing-to-pay for a marginal reduction in the risk of death.

[6] The **cost-of-illness** (COI) is the name economists give to the resource/opportunity cost-based approach to estimating the costs of morbidity. With the COI approach, as the name implies, the cost of illness is measured as the sum of: (A) the direct out-of-pocket expenses resulting from sickness (e.g. medicine, doctor and hospital bills) and (B) any associated opportunity costs (e.g. loss of earnings resulting from the sickness).

[7] When the COI approach is extended to measure the cost of premature death, it is commonly known as the **human capital** (HC) **approach.** The logic underpinning the HC approach goes something like this: while you are living and healthy your value to society is expressed in terms of the output you generate from working (please note that 'working' is very broadly defined here). The value of your labour, i.e. the wage you earn, is usually assumed to be representative of the value of your working time (and output produced). Based on this, the HC approach assumes that over your entire life the discounted expected value of your future wage stream constitutes the minimum value society attaches to your life at this moment.

Table 2. Total Damage Costs per ton of Pollutant in Rural and Urban UK (£)

	$PM_{2.5}$	SO_2-direct	SO_2 2ndary Parts.	NO_x-2ndary Parts.	NO_x-ozone	NMVOC-ozone	CO	Benzene	1.3-Butadiene
London	414768	10464	1775	1759	-820	586	3	2501	94287
Rural Oxfordshire	37228	819	1775	1759	-820	586	0	237	9374

Economic Costs of Health Effects of Vehicle Emissions: Results

The impact-pathway methodology outlined above has been applied to road (and other forms of) transport in the EU-funded ExternE project in order to estimate its' environmental costs. Applying information on population densities, emission technologies, geographical data on the road network, meteorological data and vehicle use characteristics, modelling simulations produced estimates of the health – and other environmental – costs of vehicle emissions. Summary results are presented in Table 2.

As can be seen from Table 2, the emission of NO_x via ozone leads to negative costs, i.e. benefits. This means that an additional unit of NO_x leads to a reduction in ozone and thus a decrease in ozone damages. Due to the complex non-linear ozone formation processes, for NO_x/NMVOC ratios an increase of NO_x emissions at first leads to increasing concentrations and after passing a "hill", to decreasing ozone concentrations. The UK is over the hill in this regard! It should be noted that these monetary estimates include only ozone formation on the regional scale. Local and global effects could not be quantified.

The health effects of vehicle emissions comprise 90% of the environmental costs reported in this section and are caused primarily by primary ($PM_{2.5}$) and secondary (Sulphates, Nitrates) particles. Table 3 gives the environmental damage costs per 100 vehicle kilometres for petrol cars complying with EURO2 and a cylinder capacity between 1.4 and 2 litres, and the costs for diesel cars complying with EURO2 with a cylinder capacity up to 2 litres.

The aggregate UK results of the modelling exercise are shown Figures 2, 3 and 4.

Figure 2 shows that in 1997, transport sector externalities (tailpipe emissions) were estimated at 8.7 billion Euro, or £5.1 billion, which is around 1% of UK GDP. The time-series results show that there are significant benefits, in terms of reducing road sector environmental costs, from the introduction of tighter vehicle emission standards. Indeed, the overall road transport environmental costs have declined by almost 4 billion Euro, or £2.4 billion between 1990 and 1997, even though vehicle km have risen over this same time period from 419 billion km to 464 billion km.

Table 3. UK Damage costs in £ per 100 Vehicle km, by Location, Vehicle and Fuel

London	
Petrol cars EURO2, 1.4-2L	*0.65*
Diesel cars EURO2, ‹2L	*2.62*
Rural Oxfordshire	
Petrol cars EURO2 1.4-2L	*0.08*
Diesel cars EURO2 ‹2L	*0.17*

Table 4. Percentage of UK Road Sector Damages by Location, Vehicle and Fuel

	% vehicle km	% damage
By location		
Urban	43.1	64.8
Rural	39.8	21.2
Motorway	17.1	14.0
By vehicle		
Gasoline cars	73.1	34.3
Diesel cars	8.8	7.4
Gasoline LDVs	3.0	2.1
Diesel LDVs	6.0	12.5
Rigid HDV	4.4	23.3
Articulated HDV	2.7	10.3
Bus	1.1	8.9
Motorcycle	0.9	1.2
By fuel		
Diesel	22.9	62.4
Gasoline	77.1	37.6

The results clearly show that diesel fuel use generates much higher external costs than petrol. They also show that urban traffic generates much higher external costs than rural traffic. In terms of the split by vehicle category, the results show that although cars have a very high share of the vehicle kilometres driven (›80%) they only account for 40% of the damage. Goods vehicles, especially HDVs, have a disproportionately large contribution to damages (33% of damages from 7% of vehicle kilometres).

Current Methodological Issues

The economic costs of the health effects of vehicle emissions are extremely sensitive to the unit value that is given to value of a premature death. It is therefore essential that, as perhaps the dominant single component of the cost estimates of these emissions, and therefore the benefits of emission reduction policies, this unit value has an empirical robustness. The key current research effort is therefore into how this unit value is best approximated for the emissions context. Specifically, in order to produce WTP values that are tailored to the vehicle emission context the key questions over what types of adjustments to make may be formulated thus:

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[8] Thus, damage cost estimates (or benefits) are found by simply multiplying the 'number of lives saved' or 'chronic cases avoided' by the VSL and VSC, respectively.

- Should we adjust the VOSL values for the fact that many of those affected are old and have lower life expectancy?
- Should some adjustment be made for their state of health?
- Should an adjustment be made for the risk context?
- Should some adjustment be made for a lapse of time between the exposure and the impact?

Current thinking regarding these questions is outlined below.

The importance of age and life expectancy

Most compensating wage differential and CV studies measure compensation of mortality risks for individuals who are an average of 40 years of age. Assuming that an individual is productive until 65 years of age, and will live into their 70s, this values about 25 (productive) to 35 (total) life-years lost. Deaths from exposure to air pollution typically occur at an age over 65 – at least in Western Europe – which implies much fewer life-years lost. This therefore suggests that an 'age adjustment' may be required when applying these VSL estimates to deaths related to air pollution exposure.

For example, a UK government working group on the valuation of the health impacts of air pollution (The Stationery Office, 1999), recommends the application of adjustment factors to the baseline WTP-based VSL. In recognition of the fact that epidemiological studies only report data on the whole population at risk over 65, the group suggests using an adjustment factor of 70 percent, weighted to take into account of the average distribution of ages within the over 65 age group in the general population. Further adjustments are made to reflect the life expectancy of the affected individual. The end result is a VSL ranging from £^{1996}1.4 million to only £^{1996}2,600 depending on the adjustments made to the baseline value.

Markandya (1997) directly derives the value of a life year lost (VLYL) from the baseline VSL. These estimates are then used to value the years of life lost as a result of acute mortality, where a premature death is assumed to result in 0.75 years of life lost. For example, estimates of (male) survival probabilities for the EU population obtained from Eurostat were used to estimate the VLYL for individuals aged from 35 to 45 – using a discount was in the range 0 to 3 percent and a baseline VSL of £2 million. The mid-point estimate, which we adopted in the 'ExternE' project for the 'standard' acute VLYL, was £61,624.

Pure health impairment

An important assumption underpinning the analysis above is that the person at risk is in an 'average' state of health for someone of his or her age group. Clinical evidence would suggest, however, that individuals who die prematurely from air pollution are already in a state of severely impaired health. If a person's quality of life is severely impaired this may affect his or her WTP for a reduction in the risk of death. The question therefore arises as to whether it is appropriate to adjust the VLYL to account for impaired health (or quality of life).

There is little empirical evidence however, substantiating a significant positive correlation between WTP and the quality of life that individuals expect during an additional year of life, although health service professionals do use a 'Quality Adjusted Life Years' (or QALY) approach in their decision making process.

Work has been undertaken in the UK, (The Stationery Office, 1999), which examines the effect of assuming that WTP is **proportional** to an index of health-related quality of life. Taking the loss of one year, for example, the suggested quality of life adjustment factor is 53% (±1 SD: 26% to 92%). However, the quality of life measure used in the study may not be representative of all types of respiratory illness. Also, there is no evidence as to the extent to which the quality of life might depress WTP; it is quite possible that WTP might change more or less than proportionately to changes in the quality of life. There is therefore a high degree of uncertainty associated with such an adjustment. This suggests that no adjustment should be made to VSL/VLYL estimates in this regard at present.

The context of air pollution risks

Factors influencing the nature of the risk itself may affect individuals' WTP for avoiding particular types of risks. Such factors include:

- The type of health effect (e.g. people may dread a chronic risk more than an acute risk).
- The risk context (e.g. evidence shows that people regard involuntary risks, over which they have no control, as worse than others).
- The futurity of the health effect (e.g. effects which happen today are generally worse then those which occur tomorrow).

Whilst there are very good reasons to believe that the risk context is likely to be important in determining the WTP values that we wish to derive, it is too soon to make such an adjustment for policy purposes. Empirical evidence of this effect is therefore required as a matter of urgency.

Latent and chronic mortality

If exposure to vehicle-induced air pollution today causes the risk of death to increase T years from now the WTP to avoid that risk is not likely to be the same as that associated with an increase in the risk of death now. The accepted way to deal with such latency effects is to discount future risks. Hence, if the WTP for an immediate reduction in risk is £X, then the WTP for a reduction in a risk with a latency period of T years is $X \cdot (1 + r)^{-T}$. The key question, of course, is what value should r take? The numbers derived for chronic mortality in the case of PM exposure are: £61,624 (0% discount rate) and £56,765 (3% discount rate).

Measurement of the cost per life saved (CPLS)/cost per life year saved (CPLYS)

It is well known that both the VSL and VLYL figures are higher than the values that are implicitly or explicitly used by policy makers in EU countries, and represented by the resource cost of life (year) saved. For example, values relevant to the vehicle pollution context that have been identified are given in Table 5 below.

Table 5. Values of CPLS used in Different Interventions (1998 prices)

Source	Date	Central Value (£)
UK	1997	376,765
Norway	1995	1,272,267
Germany	1997	479,111
Netherlands	1998	8824

One way to reconcile the discrepancy between the values considered is to accept that the WTP values are subject to many measurement errors and uncertainties so that they are in fact over-estimated by a factor of 10. Statistical analysis on existing data supports this possibility. Therefore, in view of this high level of uncertainty, and the reasonably open and democratic process by which investment in health is decided within these countries, it could be argued that both the scientific/economic WTP estimates and the revealed preference estimates should be used in the damage cost estimation until further empirical evidence is forthcoming.

Concluding comments on methodological issues

The discussion above reveals that, both in empirical and methodological terms, the valuation of mortality risks in the vehicle pollution context is in its infancy. It is also worth noting that all valuation exercises only capture partial benefits. For example, valuation of the health effects of air pollution using the techniques above will not capture lifestyle changes, sub-clinical physical and mental discomfort caused, or some of the more chronic intangible effects. An example of the latter is that children might not be allowed to go outside to play due to the poor air quality, and this could affect their quality of life, physical activity and have implications for long term well-being; poor air quality may also cause eye irritations which affect the ability to read.

Many of the existing studies also do not include the costs of illness in calculations of the value of premature mortality from exposure to air pollutants. In reality however, people who die from respiratory diseases often suffer prolong periods of illness, so including the costs of illness would actually give a more accurate estimation of the costs of death from air pollution.

Overall Conclusions

The paper has identified the best estimates of the economic costs of health effects of vehicle emissions. In the UK these total around 1% of GDP. The impact pathway modelling methodology used to derive the estimates is outlined. The key methodological and empirical issues that arise in making these estimates are also summarised. The key difficulty in valuing these damages appears to revolve around the valuation of premature death. Variations in this value are likely to significantly affect the aggregated damages estimated using the impact-pathway methodology and this is regarded as a priority research issue in UK and Europe at present.

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Figure 1. Key Components of Air Pollution Assessment

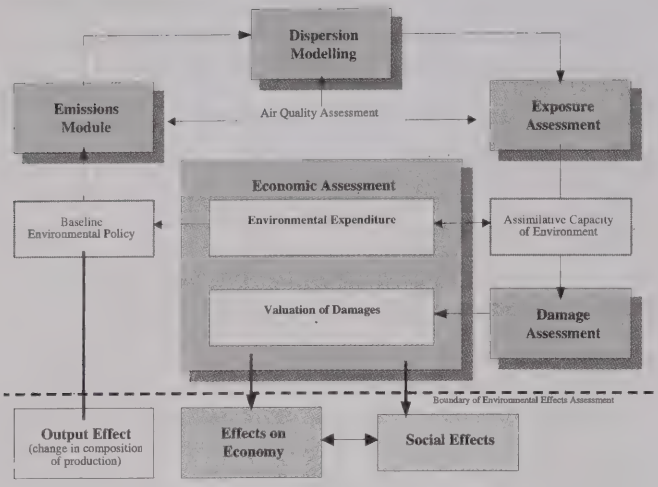
Dispersion Modelling
Emissions Module
Air Quality Assessment
Exposure Assessment
Economic Assessment
Baseline Environmental Policy
Environmental Expenditure
Assimilative Capacity of Environment
Valuation of Damages
Damage Assessment
Boundary of Environmental Effects Assessment
Output Effect (change in composition of production)
Effects on Economy
Social Effects

Figure 2. Total UK Road Externalities by Pollutant, 1990-1997

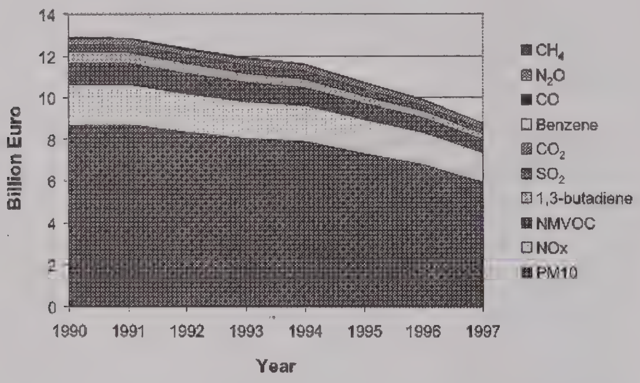

Figure 3. Total UK Road Sector Environmental Costs 1990-1997, by Vehicle

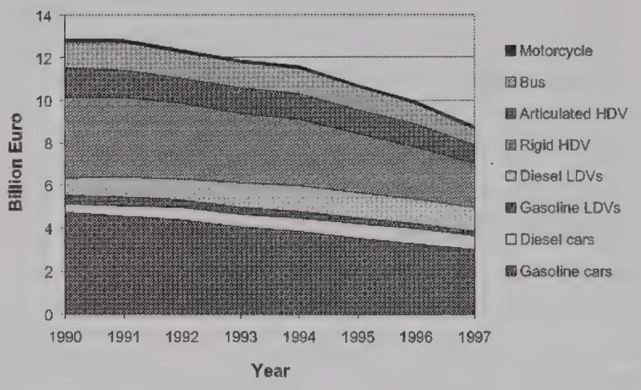

Figure 4. Total UK Road Sector Environmental Costs 1990-1997, by Fuel and Location

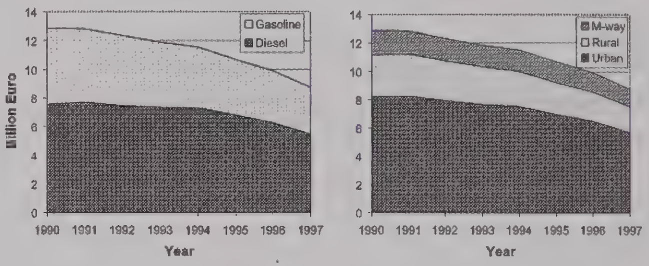

Table 6. Comparison of Unit Values Used in Several Major Studies

Health Outcomes	US EPA (1999)			US TAF			Canada AQVM (1999)			EU ExternE EC (1999)
	Low (US$ 1990)	Central (US$ 1990)	High (US$ 1990)	Low (US$ 1990)	Central (US$ 1990)	High (US$ 1990)	Low (US$ 1990)	Central (US$ 1990)	High (US$ 1990)	Central (US$ 1990)
Mortality	1,560,000	4,800,000	8,040,000	1,584,000	3,100,000	6,148,000	1,680,000	2,870,000	5,740,000	3,031,000
Chronic Bronchitis	-	260,000	-	59,400	260,000	523,100	122,500	186,200	325,500	102,700
Cardiac Hospital Admissions	-	9,500	-	-	9,300	-	2,940	5,880	8,820	7,696
Respiratory Hospital Admissions	-	6,900	-	-	6,647	-	2,310	4,620	6,860	7,696
Emergency Room Visits	144	194	269	-	188	-	203	399	602	218
Work Loss Days	-	83	-	-	-	-	-	-	-	-
Acute Bronchitis	13	45	77	-	-	-	-	-	-	-
Restricted Activity Days	16	38	61	-	54	-	26	51	77	73
Respiratory Symptoms	5	15	33	-	12	-	5	11	15	7
Shortness of Breath	0	5.3	10.6	-	-	-	-	-	-	7
Asthma	12	32	54	-	33	-	12	32	53	36
Child Bronchitis	-	-	-	-	45	-	105	217	322	-

Source: Davis, Krupnick and Thurston (1999)

[1] Low and high values are estimated to be 1 standard deviation below and above the mean of the Weibull distribution for mortality; for the other health outcomes they are the minimums and maximums of a judgemental uniform distribution.

[2] Low, central and high estimates are given probabilities of 33, 34 and 33 percent, respectively.

[3] Low and high values are the 5 percent and 95 percent tails of the distribution.

[4] Uncertainty bounds are set by dividing (low) and multiplying (high) the mean by the geometric standard deviation (2).

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CONTENTS

Editorial 67

Measurements of PM_{10} Emissions from a Construction Site – A Case Study 68
Stuart Upton and Vina Kikadia, BRE Environment

Stage 2 Vapour Recovery on Petrol Filling Stations – Is it Worthwhile? 72
David Muir, Bristol City Council

Diurnal Cycles of Carbon Monoxide and Nitric Oxides Near Schools 75
Edward Hopley (City of Norwich School) and
Peter Brimblecombe, University of East Anglia

Providing Good Indoor Air Quality in Urban Buildings – Overcoming the 'Circle of Blame' 78
Dr Vina Kukadia, BRE Environment

Developers & Planners: Delivering Waste Management Facilities through the Planning System 82
Alan Potter, Golder Associates (UK) Ltd

The Winter Edition of Clean Air is being replaced by a special edition commemorating the Great London Smog of 1952, to be published on 5 December 2002

The National Society for Clean Air and Environmental Protection produces information, organises conferences and training events, and campaigns on air pollution, noise and environmental protection issues. Founded in 1899, the Society's work on smoke control led to the Clean Air Acts. More recently NSCA has been influential in developing thinking on integrated pollution control, noise legislation, and air quality management.

NSCA's membership is largely made up of organisations with a direct involvement in environmental protection: industry, local authorities, universities and colleges, professional institutions, environmental consultancies and regulatory agencies. Individual membership is also available to environmental specialists within industry, local authorities, central government, technical, academic and institutional bodies.

Members benefit from joining a unique network of individuals who share an interest in a realistic approach to environmental protection policy; from access to up-to-date and relevant information; from reduced fees at NSCA conferences and training events. They contribute to NSCA's regional and national activities; to environmental policy development; to translating policy into practice; to the Society's wide-ranging educational programmes.

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Editorial

CLEARING THE AIR

December sees the 50th anniversary of the pollution event which has become known as the Great London Smog, an event which is commonly viewed as the watershed which gave rise to modern air pollution control in the UK and elsewhere. The Smog, which started to develop on Thursday, 4 December and dissipated in the early hours of Tuesday, 9 December, killed an estimated 4,000 people, although some put this figure much higher. A committee of investigation was assembled under Lord Beaver, which in 1954 recommended that there be a clean air act, which was initiated as a private member's bill by the Conservative MP Gerald Nabarro. Nabarro, who had been thoroughly briefed by NSCA's forerunner, the National Smoke Abatement Society, withdrew his bill when the Government of the day put forward their own Clean Air Bill, which became the Clean Air Act 1956 on 5 July that year.

All ancient history? We would certainly like to think so. However, while it is true to say that major pollution incidents resulting in mass fatalities are a thing of the past for the UK, industrial accidents and poor pollution control are still taking their toll around the world, and on a much larger scale. The accident at the Union Carbide plant in Bhopal 18 years ago killed tens of thousands, and the recurring smogs across South East Asia, seen again this year, comprehensively dwarf London's "killer" smog of 1952. The inability of the World Summit in Johannesburg to produce a useful global agreement for the introduction of renewable energy sources (naming no names) is a setback, but it is not the only mechanism. Renewable energy and clean transport surely hold the key if mass pollution incidents are to be truly consigned to history.

NSCA will be marking the anniversary of the Great London Smog with the publication of a special edition of Clean Air, which will replace the normal Winter issue. Twelve key commentators on air pollution will review the 50 years since 1952, and look forward to the next 50, assessing the prospects for truly clean air. NSCA is also holding a conference in association with the Parliamentary Office of Science and Technology (POST) at the Houses of Parliament, coinciding with its AGM on 5 December.

In this issue, we deal with issues closer to home. Two papers from BRE look at particle emissions from construction sites and at the ways in which good indoor air quality can be designed into urban buildings. David Muir, of Bristol City Council reports on the impact of Stage 2 vapour recover on benzene concentrations around petrol stations.

We are also very pleased to be able to publish work carried out under the Nuffield Science Bursary programme at the University of East Anglia by Edward Hopley, a pupil at City of Norwich School. Edward, with the assistance of Prof. Peter Brimblecombe, reports on the diurnal variations of CO and NOx near schools, information necessary to the development of action plans to reduce childhood exposure to urban air pollutants.

Finally, Alan Potter, of Golder Associates (UK), discusses two arcane subjects in one article: how to deliver the objectives of the Landfill Directive through the planning system.

REPORTS

Measurements of PM_{10} Emissions from a Construction Site: A Case Study

Stuart Upton and Vina Kukadia

BRE Environment

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Summary

The paper describes results from continuous PM_{10} monitoring conducted at three locations around a construction site, during the site preparation, demolition and earth-working phases. The site was operated to current best practice methods for controlling the emission of dust and fine particles. During the period of these measurements, no complaints relating to dust and particle pollution were received at either the site office or the local authority, resulting in an uninterrupted and thereby cost effective work programme.

Increases in the background PM_{10} concentrations were recorded at the site boundary during site operating periods. Typically these were about 3, 5 and 11 $\mu g\ m^{-3}$ during the site preparation, earth-working and demolition activities respectively. During the site preparation and earth-working phases of the development, the additional PM_{10} recorded was likely to be from the emissions of diesel engines powering the plant and trucks at the site. The earth-working phase was conducted during the winter months when the ground and materials were moist, so wheel-raised and materials handling emissions were very low.

No discernible increase in the concentration of PM_{10} could be detected at outlying monitoring stations (situated around 150 metres away) when they were downwind of the construction site. Therefore, during the types of activities monitored here, although small increases in particle concentration at the site boundary may be unavoidable, if a site is operated to best practice the impact on the wider local environment (>150 m from the site) can be negligible.

1.0 Introduction

Nuisance dust emissions from construction and other civil engineering activities can be a common problem. Fine particles within the PM_{10} fraction from construction sites are also now recognised as significant local sources of pollution. Under Part IV of the Environment Act 1995 and the UK Air Quality Strategy (DETR, 2000), local authorities are required to work towards achieving national air quality objectives. It is likely that construction site operators will therefore need to demonstrate that both nuisance dust and fine particle emissions from their sites are adequately controlled and are within acceptable limits. There are currently approximately 50,000 active construction sites in the UK at which construction work costing more than £250,000 is in hand. Since all of these sites have the potential to generate dust and fine particles, they can be a major local source of air pollution.

Currently, there is little quantifiable information on, or understanding of, the actual contribution of construction sites to local particle concentrations and the distances over which these emissions might have an effect. As part of a larger project on minimising particle emissions from construction, a site has been monitored continuously through various stages of the construction process. This paper describes the PM_{10} measurements made at the site during the site preparation, demolition and earth-working phases of the development. At the time of writing this paper, development work at the construction site is continuing, along with measurements of particle concentrations.

2.0 Description of the construction site and monitoring locations

The construction site monitored was an old industrial building (dating from about 1820), covering an area of approximately 6500 m^2. All of the buildings were decontaminated and then stripped of any asbestos, before being demolished and removed from the site. Following this, at least a 1-metre depth of soil was removed from the entire site, before the site was in-filled and levelled.

Figure 1 shows a diagram of the layout of the construction site and the locations of the PM_{10} monitoring stations. Continuous measurements of PM_{10} concentrations were taken at all three of the monitoring stations. Meteorological data was recorded at the monitoring station next to the construction site (#1). The site operator provided copies of working method statements and daily site activity records to enable comparisons with particle concentration measurements to be made. This provided information on the nature and location of activities and vehicle movements at the site.

Best practice was followed at the construction site to minimise the emission of particles into the atmosphere. This included:

- fully scaffolding, hoarding and sheeting the perimeter of the site;
- controlled "nibbling down" of the buildings with specially modified excavator grabs;
- the use of water sprays during demolition; and
- wheel-washing of every vehicle leaving the site.

3.0 Results

3.1 Daily mean PM_{10} concentration results

A summary of the mean 24-hour PM_{10} concentration data for the three monitoring stations during the measurement period is shown in Figure 2. The figure also shows the activities underway at the construction site during this period. The 24-hour daily mean concentration of the PM_{10} fraction is the only one for which a UK national atmospheric air quality standard (DETR, 2000) has been set. Currently, the objective for PM_{10} particles is at a mass concentration limit of 50 μg m^{-3}, with up to 35 exceedences allowable each year.

Figure 2 shows that, despite all of the construction activities that took place during the six month monitoring period reported here, there was only one day when the limit concentration of 50 μg m^{-3} was exceeded (9th January 2002). This exceedence only occurred at the monitoring station nearest to the construction site (#1). It is therefore very likely, although not absolutely certain, that it was associated with emissions from the site, rather than from any other local source. It should be noted that this exceedence was associated with an episode of high pollution in general, with high PM_{10} concentrations being recorded at all three monitoring stations. The combination of this higher pollution episode and the demolition activities at the construction site during the working day (0800 to 1800 hours), was sufficient to raise the 24-hour PM_{10} concentration above the 50 μg m^{-3} limit.

3.2 Effect of construction activities on PM_{10} concentrations at the site boundary

Table 1 shows the impact of the construction site on PM_{10} concentration at the site boundary, for the various phases of activity at the site. This has been estimated by calculating the difference in PM_{10} concentration between the monitoring station next to the construction site (#1) and an assumed background concentration (the mean of the other two stations, #2 and #3). All of the figures quoted in the table are for periods when the site was potentially active, i.e. Monday to Friday, 0800 to 1800 hours only.

During periods of limited activity at the site (site preparation and hoarding, scaffolding and sheeting and during the Christmas break), the mean PM_{10} concentration at the station next to the site (#1) was around 2.1 μg m^{-3} higher than at the other more remote stations (#2 and #3). This probably arises from its proximity to the High Street. If this figure is taken as a typical difference between the PM_{10} concentrations at the site boundary and background monitoring stations, the increases in concentrations at the site boundary during the various phases of the development were approximately:

Table 1. Differential change in PM_{10} concentrations at the site boundary (μg m^{-3})

Activity	**Duration of activity (weeks)**	**Mean PM_{10} difference between site boundary and background (μg m^{-3})**
Site preparation & hoarding	5	3.2
Scaffolding & sheeting	3	0.3
Soft stripping inside buildings	4	5.9
Demolition and removal of hardcore, timber etc.	8	13.6
Christmas break	2	2.8
Piling and earth-working	3	13.6*
Earth-working	5	5.0

* During part of this period the diesel generator for the piling rig was positioned only about 5 m from the PM_{10} monitoring equipment at station #1.

- around 3 μg m^{-3} during operations to strip the buildings internally;
- around 11 μg m^{-3} during demolition operations;
- around 5 μg m^{-3} during earth-working operations.

Activities likely to have caused elevated PM_{10} concentrations during the internal stripping of the buildings include: diesel engines running extract systems during asbestos stripping; plant handling and sorting the stripped materials from inside the buildings and trucks taking this material away from the site to landfill.

As an indication of the level of potential emissions from vehicle engines, during the 8-week period covering demolition of the buildings, as well as the emissions from the demolition itself and the demolition plant, over 300 trucks accessed the site to remove demolished materials.

Similarly, within the 8-week earth-working period, other than the piling and earth-working plant operating at the site, over 1400 trucks accessed the site either removing material from, or bringing material to the site. During one week within this period the diesel generator for the piling rig was positioned very close to the #1 sampling station, giving elevated PM_{10} concentrations in the immediate vicinity.

3.3 The effect of site operations on PM_{10} concentrations on the wider local environment

The distance to which emissions from the construction site can affect the wider local environment is not known, although it can be estimated using dispersion and dilution rates if the source concentration of the pollutant in question is known.

The data from the two more remote monitoring stations (at ~150 m from the construction site) have been compared, along with wind direction data, to try to identify if the emissions from the construction site can be detected at these distances. All of the comparisons made were from periods within the working week, when the construction site was active (Monday to Friday, 0800-1800 hours). Monitoring station #2 was situated about 185 m south-west of the construction site, with monitoring station #3 positioned about 130 m east of the site (see Figure 1). Two wind sectors were chosen for investigation. The first sector (70° to 110°) was chosen to broadly represent when the wind was blowing from east to west across the construction site towards sampling station #2. The second sector (250° to 290°) was chosen to broadly represent when the wind was blowing from west to east across the construction site towards sampling station #3. These two wind sectors are shown diagrammatically in Figure 3.

Winds from the easterly sector (70° to 110°) first pass over sampling station #3 upwind of the site, before crossing the construction site and then moving on towards the downwind station (#2). Any PM_{10} particles emitted from the construction site should be carried towards the downwind monitoring station (in this case station #2) and show up as a slightly increased concentration. Similarly, with winds within the westerly sector (250° to 290°), emissions from the construction site should be carried downwind, in this latter case towards monitoring station #3.

Table 2 shows the change in PM_{10} concentration between the upwind (#3) and downwind (#1) monitoring stations for the easterly wind sector (70° to 110°), broken down into the time periods covering the various activities at the construction site.

Table 3 shows the change in PM_{10} concentration between the upwind (#1) and downwind (#3) monitoring stations for the westerly wind sector (250° to 290°).

The results show that overall, there seems to be an approximately 2 $\mu g\ m^{-3}$ difference between the two monitoring stations, with station #3 being higher than station #2 most of the time. It could be surmised that this difference was probably due to the more rural location of the monitoring station #2, which was situated on a hill and on a road with access to residents only. No obvious influence of PM_{10} particle emissions from the construction site can be seen at these distances downwind (130 m and 185 m for monitoring stations #3 and #2 respectively).

Table 2. Difference in PM_{10} concentrations between upwind and downwind sampling stations for easterly wind sector (70° to 110°)

Site activity	Downwind – upwind PM_{10} concentration ($\mu g\ m^{-3}$)
Building stripping	-0.5
Demolition	-1.6
Earth-working	-3.6
Total for the entire monitoring period	-1.7

Table 3. Difference in PM_{10} concentrations between upwind and downwind sampling stations for westerly wind sector (250° to 290°)

Site activity	Downwind – upwind PM_{10} concentration ($\mu g\ m^{-3}$)
Building stripping	+4.5
Demolition	+2.1
Earth-working	+2.4
Total for the entire monitoring period	+2.3

4.0 CONCLUSIONS

Continuous measurements have been made of PM_{10} concentrations at three locations around a construction site through various phases of the development.

The site was operated to "Best Practice" and no complaints were received either at the site office or local authority during the monitoring period.

Increases in concentrations of particles in the PM_{10} fraction at the site boundary during working hours (Monday to Friday, 0800-1800 hours) for the various phases of the development were approximately:

- around 3 $\mu g\ m^{-3}$ during operations to strip the buildings internally;
- around 11 $\mu g\ m^{-3}$ during demolition operations;
- around 5 $\mu g\ m^{-3}$ during earth-working operations.

No increases in PM_{10} concentration attributable to activities at the site could be detected at the downwind monitoring stations (about 150 m away).

The findings from this work show that it is possible to control PM_{10} emissions from construction sites. To help achieve this:

- emissions need to be controlled at source;
- site method statements, techniques and emission controls need to be specified and agreed with the local authority from the outset of the project;
- the costs of any control measures need to be incorporated into all tenders for the development; and
- any PM_{10} monitoring required to be conducted at or around a construction site needs to be on a continuously recorded basis.

5.0 Acknowledgements

The work reported here is part of the Partners in Innovation project 'Minimising Airborne Particle Pollution from Construction Sites' which is in part funded by the

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- WSP Environmental

All contributions are gratefully acknowledged.

6.0 Further Information

A more detailed report on all of the monitoring conducted at the site will be available on completion of the measurements.

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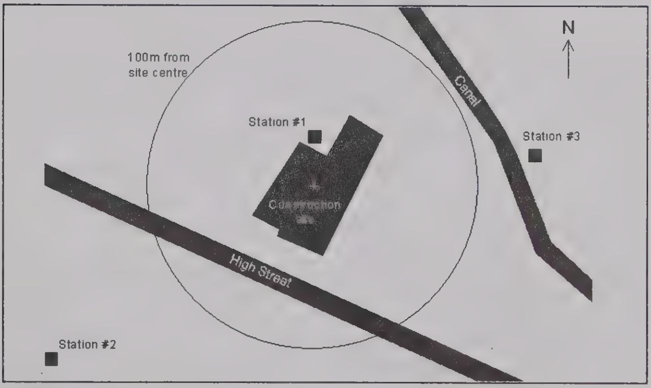


Figure 1. Site map showing location of monitoring stations and local features

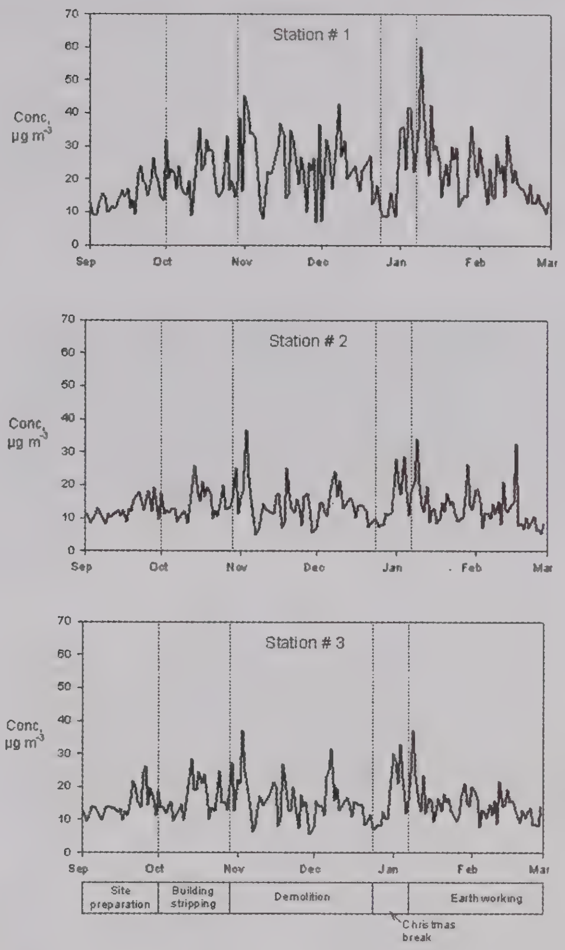
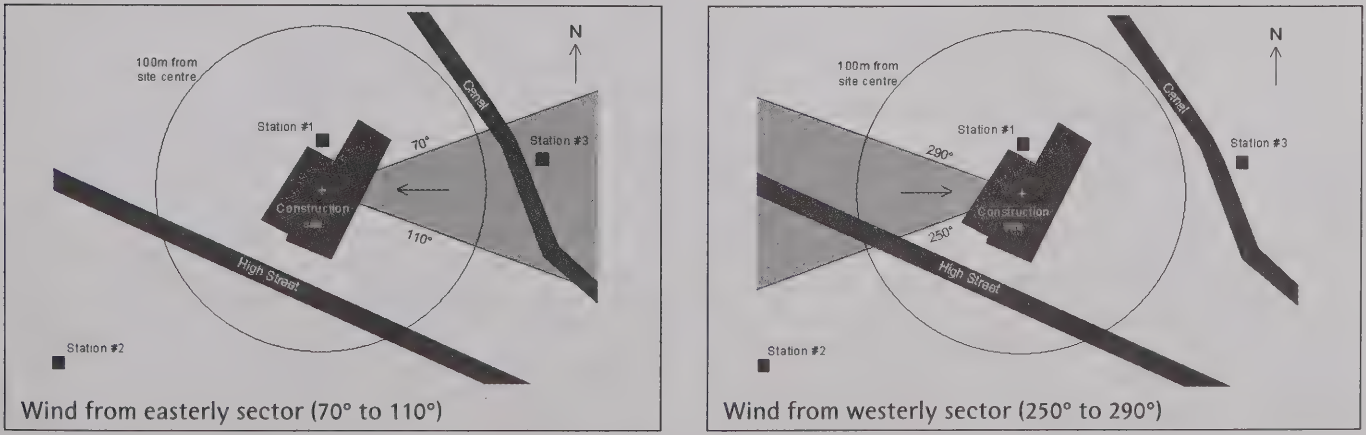


Figure 2. Summary of all 24-hour mean PM_{10} concentrations from the three monitoring stations



Wind from easterly sector (70° to 110°)

Wind from westerly sector (250° to 290°)

Figure 3. Wind sectors investigated to compare upwind and downwind PM_{10} concentrations

Stage 2 Vapour Recovery on Petrol Filling Stations – Is it Worthwhile?

David Muir

Bristol City Council

The Government opened consultation on the implementation of Stage 2 petrol vapour recovery in April 2002. The proposal is for a programme of introduction of the technology which is intended to reduce emissions of petrol vapour by drawing vapour displaced from the fuel tank of a vehicle back into the tanks of the filling station.

INTRODUCTION

Petrol filling stations have been recognised as a significant source of a range of organic compounds or hydrocarbons. One of these, benzene, is recognised as a geno-toxic carcinogen. As a result of this the UK Government set an Air Quality Objective of 5 ppb (16.25 µg m^{-3}) as a running annual average concentration for benzene. Subsequent European legislation has set a limit value for concentrations of benzene of 5 µg m^{-3} as an annual average to be achieved by 2010. The other hydrocarbons which are emitted are not recognised as having significant direct health impacts but they are, to a greater or lesser extent depending on the compound, implicated in the formation of tropospheric ozone. Apart from general evaporative losses there are two main sources of these compounds at petrol stations. The first is vapour displaced during a delivery of petrol by tanker. The second is a similar displacement of vapour during the delivery of petrol to a customer's vehicle. The first of these involves the displacement of several thousand litres of vapour but only happens a few times a month. The second only involves the displacement of tens of litres on each filling but happens throughout each day of opening. When the two are compared as totals the amounts of vapour which can be displaced should be broadly comparable.

Over the years measures have been taken to reduce emissions of benzene by reducing the maximum permitted concentrations of benzene in petrol. Until 1 January 2000 this was 5% but from that date the maximum permitted concentration was reduced to 1%. This became an issue of particular concern with the growing uptake of unleaded petrol as benzene has good anti-knock properties.

The only legislative measure which has been taken to reduce emissions of the other hydrocarbons has been the requirement for petrol stations to fit Stage 1 vapour recovery to address the problems of tanker deliveries. For petrol stations with a throughput greater than 1000 m^3 per annum this had to be in place by 31 December 1998. Where the throughput is 500 to 1000 m^3 per annum the deadline was 31 December 2001 and for smaller stations it is 31 December 2004. Stage 1 recovery entails fitting an extra pipe to the tanker creating a closed loop to recycle the vapour from the petrol station tanks into the delivery tanker.

METHODOLOGY

In 1995 a survey was started in Bristol to look at concentrations of benzene around four petrol stations and on the forecourts of two of the larger stations. This was done using Perkin-Elmer diffusive samplers exposed, so far as was possible, for periods of 2 weeks. Three of the stations were large and were fitted with Stage 1 vapour recovery either when the survey began or in its early stages. The third was a very small station (now closed) which was not fitted with any vapour recovery. At a fairly early stage of the survey a second petrol station opened near one of the monitoring sites. It later emerged that this station was at some stage fitted with Stage 2 vapour recovery.

The samplers which were located on the petrol station forecourts were attached to the main dispenser supports at a height of between 2.5 and 3 metres. The other samplers were attached to lamp posts or street sign posts at similar heights. These sites were located to the North East of the petrol stations to be predominantly down wind of the stations. At the two petrol stations where forecourt sampling took place additional samplers were placed south west of the stations to assess, so far as was possible, incoming concentrations. These off-forecourt samplers were located approximately 50 metres from the petrol stations.

OBSERVATIONS

From the earliest results it became apparent that there was a marked difference between the results from the on-forecourt and off-forecourt samplers. Apart from the expected higher concentrations of benzene, toluene and xylene, the ratios of concentrations of benzene to the other two substances were markedly different. This is shown in Figures 1 and 2. These also show that since the maximum permitted level of benzene in petrol was reduced the concentration ratios for the forecourt samplers have fallen into line with the roadside samplers.

The results from the samplers located off the petrol station forecourts have been consistently comparable with those from other roadside sites. There is no particular evidence from the concentration ratios to suggest major contributions from evaporative sources on the petrol stations. This is taken to mean that dispersion of such emissions is quite rapid although it could be confirmed by exposing a series of samplers at short intervals along a line.

The annual average concentrations of benzene, toluene and xylene are shown in Figures 3 to 5. These again clearly show the effects of the reduction in the maximum permitted level of benzene in petrol on concentrations of benzene on petrol station forecourts. They also show that at one forecourt site the concentrations of toluene and xylene have actually increased over the years.

The results from the off-forecourt sampler and those from other roadside sites operated by Bristol City Council show that this reduction also had a beneficial effect on more general concentrations of benzene. This is confirmed by results from the automated sites operated in the Hydrocarbon Network.

In 2002 the sampler near the smallest petrol station, which had by then closed was replaced with one on the forecourt of the larger petrol station with Stage 2 vapour recovery. Although only a few samples have been analysed these show clearly (Figure 6) that the concentrations of all three substances on the forecourt of the petrol station with Stage 2 vapour recovery (Forecourt 3) are substantially lower than on the other forecourts although they are not as low as the comparable off-forecourt concentrations.

DISCUSSION

This monitoring has shown that substantial reductions in concentrations of benzene on petrol station forecourts have been achieved by the reduction in the maximum permitted concentration of benzene in petrol. It also shows that this reduction has not been matched by reductions in concentrations of toluene and xylene and that there have actually been some increases in concentrations of these two. There is no immediately obvious reason for this increase at one station but not at the other. These two stations are owned by different companies. One possibility might be that one supplier increased the amounts of toluene and xylene to compensate for the reduction in benzene. Another might be that it is a natural consequence of the initial feedstocks. From the point of view of direct health implications this is probably desirable but all three of these substances are implicated in the formation of tropospheric ozone (RG Derwent *et al*, 1996 and 1998) as are the other hydrocarbons which are emitted with benzene, toluene and xylene but not measured. Given the growing suspicion that the health effects of tropospheric ozone may be greater than previously thought this is clearly a serious matter.

The early results from the petrol station forecourt where Stage 2 vapour recovery is fitted suggest that this will have a major impact on the emissions of all of the ozone precursors from petrol stations.

CONCLUSIONS

Although substantial improvements in concentrations of benzene have been achieved on petrol station forecourts by measures already introduced these have not affected concentrations of other substances favourably.

Stage 2 vapour recovery offers a major opportunity to reduce emissions of a wide range of tropospheric ozone precursors.

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The views expressed in this paper are those of the author and should not be taken to represent the views and policies of Bristol City Council.

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Figure 1. Annual Average Benzene: Toluene Concentration Ratios

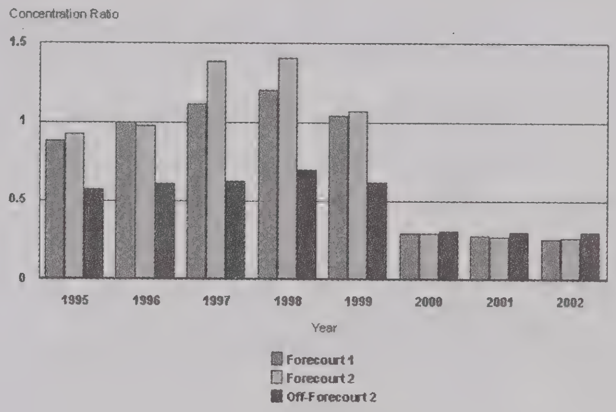


Figure 2. Annual Average Benzene: Xylene Concentration Ratios

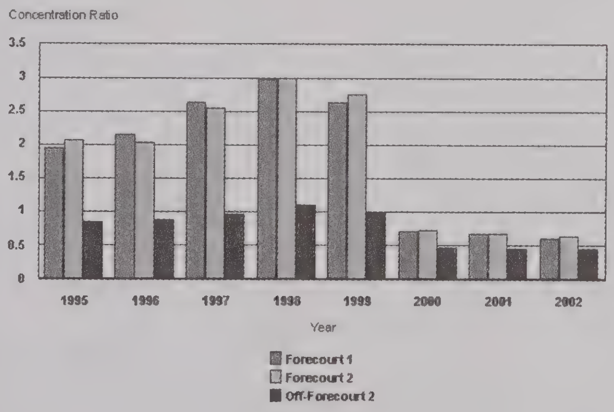


Figure 3. Annual Average Benzene Concentrations

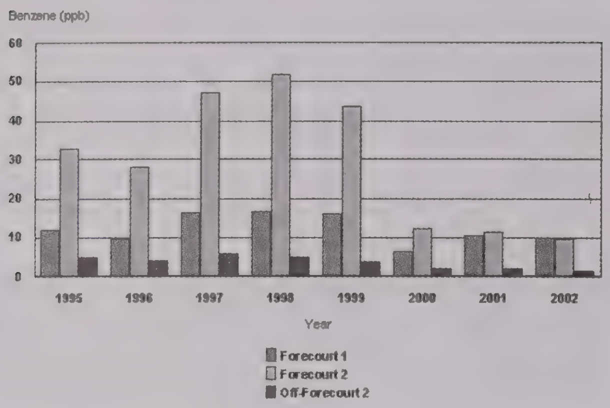


Figure 5. Annual Average Xylene Concentrations

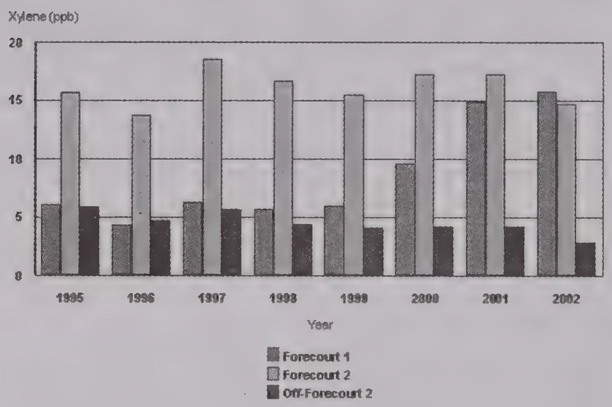


Figure 4. Annual Average Toluene Concentrations

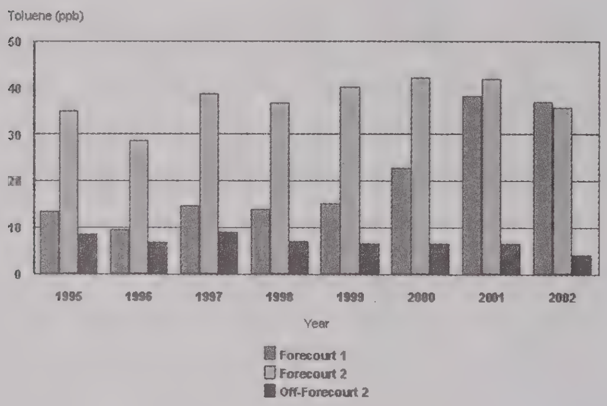
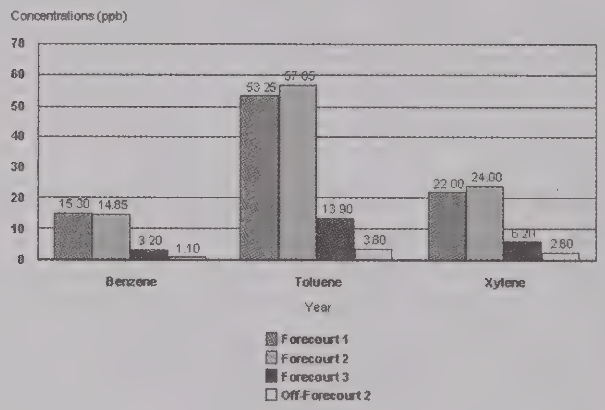


Figure 6. Average Concentrations



Diurnal Cycles of Carbon Monoxide and Nitric Oxides near Schools

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This article represents work done as part of the Nuffield Science Bursary programme in the summer of 2002. It illustrates what can be achieved by a student from a school working for a short period over the summer. This study at the University of East Anglia sought to understand an air pollution issue of importance to local communities.

Studies of the health effects of air pollution, and the resultant legislation, need to consider sensitive groups. Children are among these and their exposure to air pollutants while at school has generated a large amount of research. Much of this relates to the indoor environment, but it is also clear that children are exposed to outdoor air while at school[1, 2] and travelling each day. In the US there has been a debate over the school bus programmes which use diesel-engined buses that expose children to high levels of particulate matter[3]. There is increasing interest in the reasons for the choice of school transport mode and its impact on air pollution[4].

A rising number of children are driven to school in the UK[5], which has provoked a wide range of concerns about its social and health impacts. Exposure to air pollutants continues to be an issue that worries parents. They often believe that by driving children to school they lower the potential exposure, although this may not be true. As more children are driven to school congestion near schools increases and there is also the potential for increased emissions of pollutants as vehicles idle in wait-zones near schools. However, in the UK high traffic density near schools also raises some questions relating to the siting of air pollution equipment. School grounds represent an apparently suitable site for urban background monitoring, because instruments can often be situated some distance from the road and set in relatively secure locations. However, school traffic patterns could distort the record in school grounds. Such questions seemed to make it useful to study the diurnal cycles of some primary combustion pollutants derived from motor vehicles near schools.

METHOD

We wanted to see if the readings from air pollution monitoring stations in schools were affected by parents taking children to and from school by car. We expected that this would have some effect on the diurnal cycle of concentrations of automotive emissions, such as CO and NO_x. In particular an increase in air pollutants during the afternoon rush hour beginning about 3 pm near schools seemed a likely result coinciding with the arrival of parents in cars. This would be earlier than the overall morning rush hour, but possibly larger if cars idle while parents wait for children.

We first examined the DEFRA website by looking at the database of all their air quality monitoring stations, particularly information as to location, gases recorded and length of the data record. The initial site choice was based on the following conditions:

- The instruments were located at a school site
- They recorded the two gases of most interest, i.e. CO and NO_x
- There were no unusual features, such as a nearby industry or motorway that could have affected results.

We chose:

- **Breckon Hill School:** The School has approximately 400 pupils, ages 3-11. The monitoring station is within a self-contained, air-conditioned housing. The site is located in a residential area approximately a mile south-east of Middlesbrough town centre. The surrounding area is generally open and comprises an urban background setting.
- **Kingsbury High School:** The School has approximately 1500 pupils, ages 11-16. The monitoring station is within an existing building located within the grounds of a local school. The nearest road is approximately 30 metres from the station with an average daily traffic flow of 8,000 vehicles per day. The surrounding area is generally open and comprises school playing grounds and residential dwellings. There are a number of retail and light industrial units located within the vicinity to the east and west of the monitoring station.

The two schools chosen are from contrasting urban areas. Breckon Hill School is located in the suburbs of Middlesbrough, which is a small urban area compared to Kingsbury School, which is located in London – obviously a much larger urban conurbation. The data sets are reasonably long (i.e. 1997-2001) and allow reference back to previous years to see if pollution at the schools has increased or decreased.

We also needed to look at sites similar to the school sites, but without a school nearby. These are not necessarily easy to choose, because ideally one would want to look at a large number, or be sure that they were similar to the school sites. In the end we chose only three (Bexley, London; Princes Street, Edinburgh; and St Mary's Place, Newcastle), that were typically in suburbanised or residential areas and not near to any major roads or industry, which would affect the results.

This raw data downloaded from the website was analysed in Excel. The Excel file was flagged so weekend and weekday data could be distinguished and individual columns of hourly data averaged and sorted. The procedure was made efficient by creating a template sheet with pre-existing formulae so that new data could be simply copied and pasted in. Weekdays were taken to represent school days and weekends non-school days, although a more sophisticated analysis would have to determine whether these were indeed days on which the school was in use.

RESULTS

Figure 1 shows the differences in CO concentrations on weekdays (school days) and weekends (non school days) at the two schools. At Breckon Hill (Fig. la) the major differences between the weekdays and weekends is a much larger peak at 9.00 am on the weekdays than on the weekends. The other difference is that the afternoon peak differs in size and in the time it appears. On the weekdays it appears to rise earlier and higher. This could have been caused by parents collecting children from school after 3.00 pm thereby making the rush hour start earlier than normal for an urban area. The weekday peak although earlier and a higher total, falls off at a similar rate to that found for weekend days.

The diurnal cycle of carbon monoxide is quite different at Kingsbury School (Fig. lb). The morning weekday rush hour is very distinct and it is much lower in the weekends. The second peak is much larger in the weekends and occurs rather later in the afternoon than at Breckon Hill for all days of the week. Kingsbury is located in a large urban area and is probably more affected by the general pollution arising from a wide area, which may explain less distinctive evidence of increased carbon monoxide concentrations when children are collected from school. The fact that children at Kingsbury are older may also mean that they are more likely to make their own way to school.

At the non-school sites the averaged data indicates a diurnal range for CO that seems smaller than at the school sites (see Fig. 1c). Concentrations are quite high, but the peaks are not so well distinguished from a broad daytime maximum. Although it is hard to know if these sites are truly representative, the rush hour peaks are notably more pronounced near the two schools.

Figure 2 combines data for a number of years for both NO_x and CO at Breckon Hill School, which was a distinctive school site. The concentrations of NO_x showed the same sort of trends as was the case in the CO data with the afternoon peak rising earlier and to higher values on weekdays than during weekends. The time difference of the afternoon peak at the weekend and on weekdays was only an hour. The results in general were not quite as clear for NO_x as with the CO, but this may arise because NO_x is partially the product of oxidation reactions in the atmosphere.

It is quite noticeable that air pollution seems to have dropped uniformly from the earlier 1997-1999 to the 2000-2001 period (this was also noticeable at Kingsbury School). This decline is indicated by the consistent difference between the two lines found both for weekends and weekdays. The uniformity extends across the afternoon peak at Breckon Hill, which does not provide evidence for an increasing proportion of pollution arising from parents collecting children from school over this period evident from behavioural studies (e.g.[5]).

The overall decline in the concentration of automotive pollutants may hint at some improvement in air quality. A possible cause would be that more cars are being fitted with catalytic converters, which reduce emissions of CO and NO_x(6). There could of course be other reasons, such as improved public transport in the area reducing the number of cars, although the uniformity of the decline did not strongly support this. We might have expected stronger decreases at times of peak traffic flow in this case.

CONCLUSIONS

There is evidence that schools in less polluted urban locations may experience a different diurnal cycle of air pollutants than the general urban background. The rush hour peaks are possibly more pronounced than at typical urban locations. The characteristic early rise in the afternoon peak is more noticeable in the case of CO, compared with the reactive pollutant NO_x. We attributed the earlier afternoon rush hour to parents collecting children from school. There were general improvements in air quality with respect to CO and NO_x over the last five years, perhaps an indication of the increasing role played by catalytic converters in reducing automotive emissions. The number of sites examined in this study was very small, but the question of generation of air pollutants from school transport is important. A study, which measured the air pollution near a school and attempted to relate it to diurnal traffic flow might give a better indication of the significance of school transport to the exposure of children and the suitability of school grounds as urban monitoring sites.

ACKNOWLEDGEMENT

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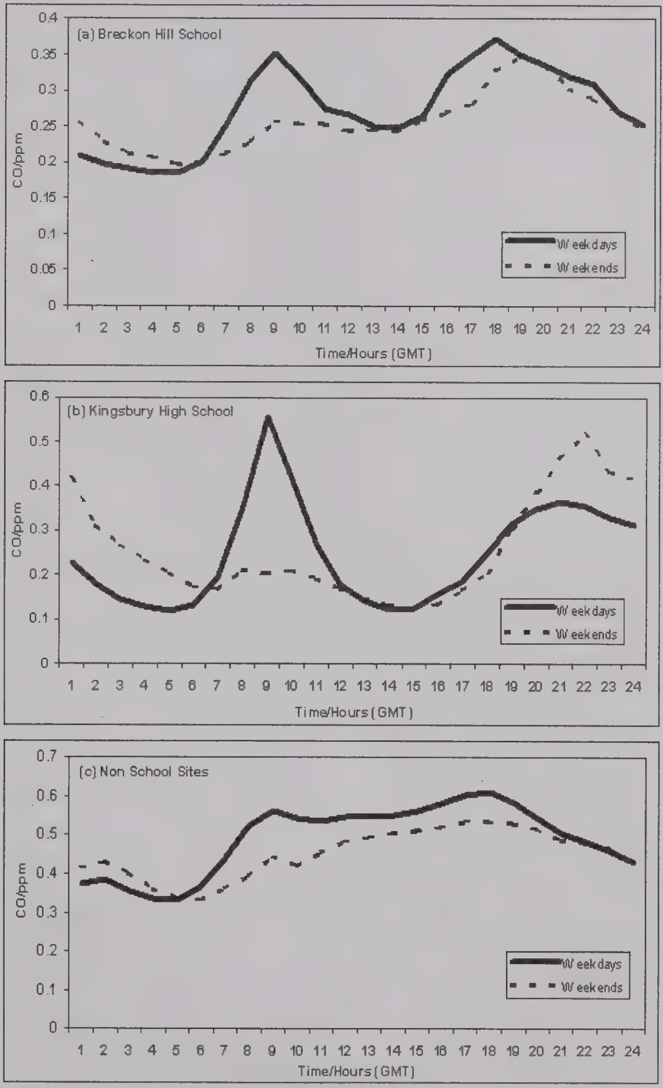

Figure 1.
Diurnal profile of mean hourly CO concentrations for 2001 on weekdays and weekends at:
(a) Breckon Hill School
(b) Kingsbury High School
(c) Average of 3 non school sites in London, Newcastle and Edinburgh

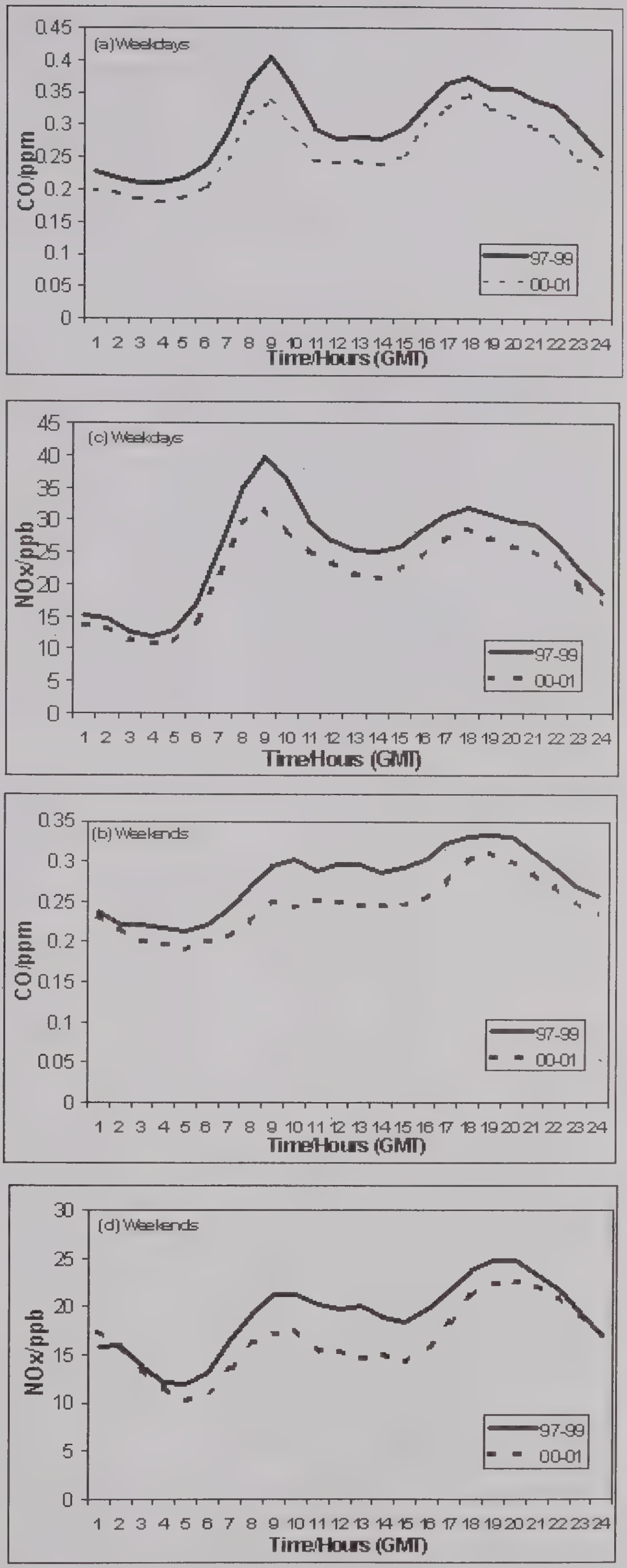

Figure 2.
(a) Mean diurnal profile of hourly CO concentrations on weekdays for the years 1997-1999 and 2000-2001
(b) Mean diurnal profile of hourly CO concentrations on weekends for the years 1997-1999 and 2000-2001
(c) Mean diurnal profile of hourly NO_X concentrations on weekdays for the years 1997-1999 and 2000-2001
(d) Mean diurnal profile of hourly NO_X concentrations on weekends for the years 1997-1999 and 2000-2001

Providing Good Indoor Air Quality in Urban Buildings – Overcoming the 'Circle of Blame'

Dr Vina Kukadia

BRE Environment

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SUMMARY

In recent years, air quality has become an important issue, in terms of both outdoor air and indoor air. It is therefore important to design buildings to minimise the ingress of external pollution into the indoor environment and to ensure that the outdoor air is not further compromised due to pollution emissions from buildings. This paper gives an introduction to some of the issues faced when designing urban buildings, in relation to outdoor pollution, together with a summary of work that is being done at BRE to address these issues.

INTRODUCTION

Historically, the improvement in urban air quality throughout the United Kingdom has been dramatic. The smogs, such as the one of winter 1952 which led to the premature deaths of 4000 Londoners, have been eliminated. This success has largely been attributed to the Clean Air Acts of 1956 and 1968, the move away from domestic coal burning and the centralisation of electricity generation in large power stations away from towns and cities.

More recently however, increases in pollutant emissions from transport, commercial and industrial activities, have led to urban air quality deteriorating once more. Air quality maps for London show how some areas of Central London suffer almost 'double the recommended limits for pollution' (SEIPH 2000). Even with the substantial financial investment in exhaust emission controls, the declaration of Air Quality Management Areas (areas where air quality standards have been exceeded) and the move to improve air quality in these areas (DETR, 2000), it could be many years before the outdoor air is of acceptable quality.

Furthermore, since people typically spend 90% of their time indoors, there is growing concern over human exposure to externally generated pollutants found indoors and their adverse impact on health, productivity, comfort and well-being. It is therefore important that pollutant ingress into buildings is minimised by effective design and operation to give good indoor air quality (IAQ). IAQ problems are generated not only from outdoor pollution sources, but are also dependent to a large extent on internally generated pollutants, such as those from furnishings, occupant metabolism and activities. Ensuring good IAQ therefore requires an understanding of the following:

- outdoor air pollution sources;
- indoor air pollution sources;
- the interaction of outdoor and indoor pollution sources with ventilation requirements; and
- the effectiveness of air pollution and ventilation control strategies.

In this paper, only outdoor pollution sources are considered.

It is also important that emissions from building use (for example, from boiler and combined heat and power discharges) and from construction and demolition activities do not further compromise outdoor air quality, as this in turn is likely to have an effect on IAQ in the building from which the emissions originate and in other buildings. The National Society for Clean Air (NSCA) Pollution Handbook, 2000 indicates that planning applications will increasingly be refused for new developments and refurbishment schemes that are likely to impact on the local environment by increasing external pollution loads (NSCA, 2000). This could be via increased emissions from the use of the building, or from vehicles related to the occupation and use of the buildings, or simply from the shape and size of the development and its effect on the pollution dispersion processes in the local region.

With the development of brown field sites, the erection of taller buildings in urban areas and the declaration of 'Air Quality Management Areas', air quality is becoming an even more important issue for all stakeholders concerned. However, if designed carefully, building developments (new and retrofits) can improve the local external environment and hence IAQ and occupant health.

AIR QUALITY AND THE 'CIRCLE OF BLAME'

The provision and maintenance of acceptable air quality is a responsibility which needs to be widely shared and can only be achieved with the collaboration of all relevant parties involved in the building specification, design and construction process. These include, for example, developers, letting agents, designers, services engineers, planners, environmental health officers, contractors and occupiers. Currently, there is a need for greater understanding amongst these parties of the integration of local environmental and air quality issues with building design. The reason for this is that, until recently, little information has been available on how pollutants disperse in urban areas, their subsequent ingress into buildings and ventilation requirements. Therefore, an effective dialogue on air quality issues does not seem to exist between the various parties. The concept of the 'Circle of Blame' has been described (Kukadia and Dimitroulopoulou, 2002), where different sectors of the industry and market blame each other and therefore do not take ownership or responsibility for a particular problem (Figure 1). In this case, the problem is the provision of good air quality, both indoors and outdoors. This process often leads to an adverse impact on the IAQ and local environment and hence delays in planning approvals, badly designed buildings and occupant dissatisfaction.

EXTERNAL POLLUTANT SOURCES AND EXPOSURE OF BUILDINGS

A wide range of air pollutants are generated outdoors and are either known or suspected of causing adverse effects to human health and the environment (DETR, 2000). Vehicular traffic has been identified as probably the largest local pollution source in urban areas (QUARG, 1993). In addition to air pollution from vehicles, other significant sources of urban pollution include local and regional industrial pollution, for example from industrial plant, boiler flues, incinerators, combined heat and power (CHP) plant and ventilation discharges.

At a particular site, pollution sources from different distances combine to bring about the overall level of pollution (Hall et al, 1997). The relationship between these various pollution sources from different distances is an important factor in the type of pollutant exposure occurring in and around buildings. For example, the nearer a pollutant source, the greater the variation in concentration over time and distance.

Principally, pollutants found in urban areas are from three main regions (Figure 2) as follows:

- **'Short-range' sources.** These are pollutants from sources at distances of ‹500 m, especially those within line of sight. Examples include pollution from vehicle exhausts, combustion, standby generators, process plant discharges, ventilation discharges, construction, demolition and nuisance sources, such as cooking smells from kitchen extracts. There can be large variations in concentration over the building, with high peak values found on building facades, and very rapid short-term fluctuations in space and time, over periods of seconds. There may also be a slower, more diffuse component of the exposure, at time scales of minutes, on one or more facades of the building.
- **'Intermediate-range' sources.** These are pollutants from sources at distances between 500 m and 10 km, usually from within the urban area. Sources are likely to be the same, as those listed above but, because rapid vertical mixing and lateral spreading occurs, concentration fluctuations are reduced and they tend to envelop the building in relatively uniform concentrations of pollutants.
- **'Long-range' sources.** These are pollutants from sources at distances ›10 km, usually beyond the urban area. Pollutants from these sources have mixed and spread over large areas and hence envelop the building in a uniform concentration of the pollutant which varies only slowly with time periods of hours to days.

The divisions between these three regimes are not sharply defined (Figure 2). The proportionate contributions of the three components to the total exposure of the building can vary considerably in different circumstances and for different pollutants. Depending upon the dominant components, the nature of the building's exposure to pollutants can vary considerably and therefore needs to be considered carefully when designing buildings for good IAQ (Kukadia et al, 2000).

BUILDING CONTAMINATION PROCESS

Internal contamination of buildings from external pollution sources depends upon both the pollutant dispersion processes around the buildings from the various sources, as described above, and the ventilation strategy used. These processes are complex and affected by the surrounding structures, as well as by the size, shape and the orientation of the building itself. The type and concentration, position and distance of the contaminant sources to which the building will be exposed are also important parameters. The arrangement of the urban area, the distribution of contaminant sources within it, the meteorology (wind speed, wind direction, weather patterns), permeability of the building envelope, indoor pollutant sources, pollution depletion mechanisms and ventilation strategies used (Philips et al, 1993, Yocum, 1982), are therefore important factors in the contamination process and need to be taken into account when designing buildings. A comprehensive review of air pollution in urban areas and building ventilation issues may be found in Kukadia et al (1999).

INDOOR AIR QUALITY IN URBAN BUILDINGS

Although concentrations of external pollutants are usually lower internally than in the external environment, cross-contamination between ventilation exhausts and inlets or contamination of inlets from stack discharges, may account for higher internal concentrations of certain pollutants than those found externally (Kukadia et al, 1996). Figure 3 shows results from building monitoring studies carried out by BRE in two buildings, one naturally ventilated (by openable windows) and the other mechanically ventilated (intake at the tenth floor level) located on a major traffic route in a city centre.

In general, external concentrations of the pollutants were higher than the internal concentrations although the latter followed the outdoor pollutant trends. The high peaks in the external concentrations were not seen in the internal environment, indicating the ability of the building to attenuate the external pollutants.

The most prominent feature of the data is the exceptionally high levels of nitrogen dioxide (NO_2) in the mechanically ventilated building during the period between the start-up and shut-down of the building's ventilation system. Levels were well in excess of the external concentrations and not seen inside the naturally ventilated building. They exceeded the UK Air Quality Strategy objective for one-hour mean for NO_2 of 105 ppb. NO_2 at such high concentrations is probably drawn into the ventilation inlet from high level boiler plant discharges. There are a number of boiler flues close to the building which may have been responsible for this, if not those on the building itself. These findings highlight the real danger of contamination of ventilation inlets at roof level from nearby sources. Thus, it is important that designers take into consideration all pollutant sources, both at ground level and those that are elevated, when designing buildings.

Recommendations on ventilation strategies for minimising the ingress of pollution are now becoming available. Apart from the effective location of intakes, these include control of intakes during higher pollution episodes; mechanical treatment of ventilation air (such as filtration); and ensuring that the building envelope is airtight. However, it is not known how effective these strategies are, as they consider local pollution sources only and do not take into consideration the overall urban pollution burden that buildings may experience.

CONCLUSION

Currently, there is a need for greater understanding of the integration of local environmental and air quality issues with building design amongst the various sectors of the industry (for example, developers, letting agents, designers, services engineers, planners, environmental health officers, contractors and occupiers). This is compounded by the fact that there is currently no comprehensive guidance for these sectors to raise their awareness of air quality and building design issues, although there is much technical guidance available. Furthermore, there is no guidance on which sectors should be included at which stage of the design process to deal with the issues. Therefore, at present an effective dialogue on these issues does not seem to exist between the various parties during the initial design stage and the issues of air quality seem to come to light only at the planning stage.

To ensure that air quality issues are considered at an early stage, and to prevent planning delays, requires the various sectors of the industry that are part of the design and construction process to be aware of all the relevant issues and to consider them appropriately early on.

ONGOING WORK AT BRE

BRE has received funding from the DTI to raise the awareness and understanding of air quality issues within the various sectors of the construction industry, by identifying and proposing ways of overcoming the barriers that inhibit the provision of good IAQ in new and refurbished buildings. The project aims to improve the process of procurement, design, construction and commissioning by developing and promoting integrated protocols for use by all key players in the process. It is intended that the protocols will provide information on air quality issues and building design and encourage collaboration between the different sectors, thereby enabling the smooth running of the building specification, design, planning and construction process whilst reducing their impact on the local environment.

A Steering Group representing developers, planners, designers and specifiers, facilities managers, environmental health officers and property surveyors, is providing guidance and practical advice to the project and will assist with the development of the protocols.

WIDER INDUSTRY INPUT

Individual focus group discussions are being held for the different sectors within the industry to determine their views on:

- how effective the construction process currently is in tackling issues of IAQ and the external environment; and
- the technical, commercial, financial and regulatory barriers that the industry faces in considering air quality and how these can be overcome.

Examples of good practice in minimising external air pollution and providing good IAQ will also be identified, analysed and promoted to the industry at large. These sessions will allow industry views to be taken into account when developing the protocols.

If you would like to contribute your views to the project, please contact Dr Vina Kukadia at BRE: Tel: 01923 664878. Fax: 01923 664443. E-mail: kukadiav@bre.co.uk

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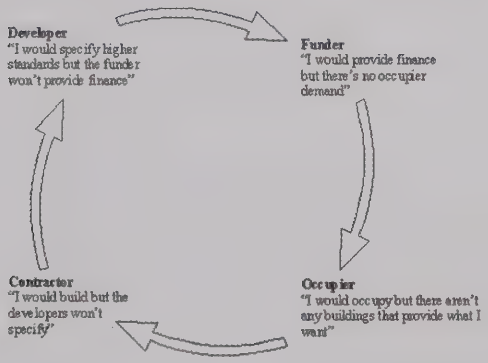
Developer
"I would specify higher standards but the funder won't provide finance"
Funder
"I would provide finance but there's no occupier demand"
Occupier
"I would occupy but there aren't any buildings that provide what I want"
Contractor
"I would build but the developers won't specify"

Figure 1. The 'Circle of Blame' Concept

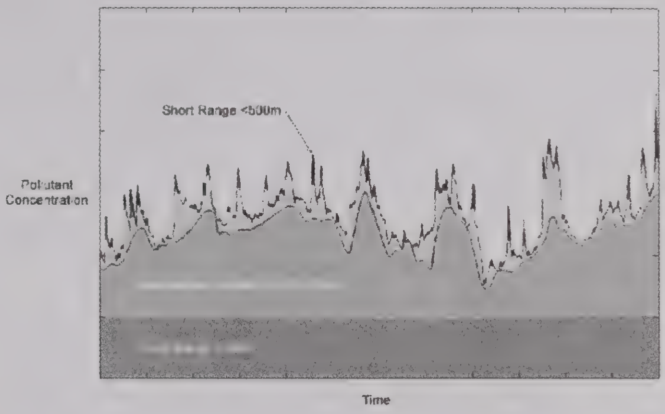

Figure 2. Pollution Contributions from Different Distance Regimes. Hall et al (1996)

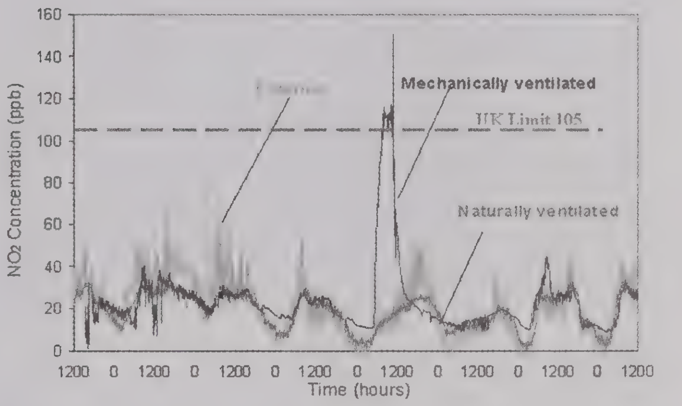

Figure 3. Air Quality in Two Buildings in a City Centre

Developers & Planners: Delivering Waste Management Facilities through the Planning System [1]

Alan Potter

Waste & Resource Management Group
Golder Associates (UK) Ltd

INTRODUCTION

With the adoption of Waste Strategy 2000 and the implementation of the Landfill Directive the demands and pressure placed on the planning system will only increase. Planning authorities are asked to make finely balanced judgements on complex decisions against a backdrop of ever increasing public scrutiny. The complexity and uncertainty associated with decisions is increasing as emerging technologies are developed and proposed and the concept of integrated waste management facilities gathers momentum. The rise in recourse to judicial review, the High Court and invoking of Human Rights legislation makes the need for decisions to be demonstrably defensible when subject to public scrutiny critical.

STEP 1: UNDERSTANDING THE PROPOSAL

There are many types of waste management facility. Each type may present distinctive issues associated with specific activities as well as generic issues. To understand the impacts associated with different facility types, the waste type and facility size needs to be considered.

Waste Type

Different waste types present different risks of impacts. The wastes can be grouped by origin or characteristics. For example commercial and industrial waste may include putrescible canteen waste and hazardous materials as well as relatively clean packaging waste. In terms of impact it is both the characteristic and the source that can have effect since it will influence the method of delivery as well as the behaviour of the waste on the receiving site. The condition in which the waste is delivered to a site will also have an influence – e.g. containerised, bagged, loose, sorted.

Facility Size

Each type of facility makes generic demands on the environment regardless of size. However it is often size and the capacity of the local environment, that determines the real burden placed on the environment in any particular case. Facilities of a capital-intensive nature may require substantial minimum tonnages to be economically viable. However emerging technologies such as anaerobic digestion can provide greater flexibility. In-vessel systems can be developed on a modular basis and there is a move towards smaller Energy from Waste (EfW) plants that serve more local catchment areas. Sizing will also be affected by the availability of suitable sites.

STEP 2: SCOPING OUT THE ISSUES

Material considerations that have a bearing on waste management planning decisions can be grouped as physical/land use issues, and public perception issues.

Land Use Issues

Waste management as a land use by its very nature presents generic impacts such as traffic movements or land take. Whilst these impacts are variable depending on the size and nature of the facility they would be common to any major development regardless of its nature. The next tier of impacts is largely attributable to the nature of the waste being received and managed on a particular site. For example putrescible waste has the potential to give rise to odour, attract vermin, flies and scavenging animals and may generate pathogens. The next tier of impacts relates to the particular type of facility. For example landfill gives rise to landfill gas which is unique to the landfill process. The size of the facility will also have an important bearing particularly with respect to the actual inputs to a site, and the site design capacity. It is the nature of the waste, the facility type and facility size that largely determine the source term of any risk factor. The final tier of impacts is the conditions within and surrounding the particular facility. This defines the risk potential and pathway and is the point at which mitigating measures primarily come into play.

Thus the following hierarchy can be arrived at to classify the land use issues:

- Generic
- Waste type specific
- Facility type specific
- Facility size specific
- Location specific

The negative and positive impacts of specific waste management facilities can be weighted when determining the environmental sustainability through the identification of criteria by which the significance of the different impacts can be assessed such as:

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[1] This article is based on a framework document prepared with the input of the following experts: Trevor Lindley, Bob le Clerc, Mike Brown, Phil Longhurst and Peter Radmell. Acknowledgements are due to all these contributors.

- Sphere of influence or 'footprint', e.g. the wider impact spatially.
- Duration, e.g. are impacts limited to a particular phase in the facility life?
- Reversibility, e.g. the wider impact over time.
- Potential for cumulative effects, e.g. account for carrying capacity of the site.
- Indirect and secondary considerations, e.g. property values, services capacity.

Facility Location

Inevitably, different facilities have different siting requirements as well as making different demands on the immediate environment and infrastructure. Facilities such as landfill are relatively fixed in location – in relation to the presence of quarries or despoiled land with suitable geology away from aquifers requiring special protection (or further if the Agency's proposed landfill location policy is adopted). Land raise provides a greater degree of flexibility but requires a significant area of land that is sited away from sensitive receptors and within a landscape that can accommodate it. In contrast Material Reclamation Facilities (MRFs) can be sited within a range of locations for example in converted council depots sited in urban areas, e.g. Kensington & Chelsea, based at existing landfill sites – generally semi-rural/rural – e.g. Mason's near Ipswich, and sited on industrial estates – generally urban fringe – e.g. Colnbrook near Heathrow.

Siting of a range of facilities including MRFs, e.g. Masons, and EfW sites, e.g. Capel in Surrey, on existing landfill sites is a growing trend which offers distinct benefits of not having to go to greenfield sites although it can extend the duration of amenity issues such as continued traffic beyond the expected life of the landfill. Location in turn will influence size via land availability, and infrastructure capacity which in turn will have a bearing on the burdens associated with transport of waste.

There is an interplay involving the three variables to identify the optimum size and location for a particular facility within the development constraints that exist in any given situation.

Public Perception

Almost all proposals for waste management facilities encounter public opposition. Waste management facilities are generally regarded as 'bad neighbour' development and the importance of analysing the significance of the public's concerns and extent to which they should be taken into account cannot be underestimated.

Public attitudes towards waste management facilities have a major influence on the progress of applications through the planning and pollution control permitting process. Not only can concerned members of the public object to proposals and influence their passage, but if sufficient weight is gained, can bring pressure to bear on elected councillors – who act as decision makers at a local level – to the extent that applications for permission are refused, even against officer advice. The widening of public consultation procedures for pollution control permitting is opening that process – which has previously been essentially technical – to similar pressures.

In addition to public concerns about primary impacts there is a precedent towards an acceptance that the fear of potential impacts alone can be regarded as a material consideration in a planning decision. This started in 1997 with the High Court judgement Newport BC v Secretary of State for Wales, which involved the proposed construction of a chemical waste treatment plant. The proposal gave rise to substantial public opposition. One of the LPA's reasons for refusing permission was that it was "perceived by the local community to be contrary to the public interest generally and their interests in particular."

> The Court of Appeal held that the perceived fears of the public were planning considerations capable of amounting (perhaps rarely) to a good reason for refusal of planning permission. That being so, the inspector should have accepted that the perceived fears of the public, even though they were not soundly based upon scientific fact, were a relevant planning consideration. He should have then gone on to consider what weight to give that concern..." *[Brady M. (1999) material considerations. Planning Inspectorate Journal, issue sixteen, autumn 1999]*

This means that planning authorities and developers need to understand the causes of public concern and the planning authority must be capable of gauging the appropriate weight to attach to these concerns. The processes by which these concerns develop are subtle and often deep-seated and usually will not simply be resolved by reducing the physical impacts of a facility. The public finds it hard to grasp the concept of risk assessment since the implication is that there is an element of risk associated with proposals, as there is with most activities in life. Whilst this is accepted and understood by waste management professionals, the public still seek assurances of absolute safety.

The public must be engaged in the decision making process, educated about the options available and the true impacts of each and be given the opportunity to influence the development and operation of the facility if permission is granted. The Human Rights Act, the Aarhus Convention and other international obligations, which require the planning system to be increasingly responsive to the public's wishes and concerns, reinforce this.

The Government's stated commitment to public participation in the planning system is set out in the recent Green Paper on the reform of the Planning System which proposes that the "Local Development Framework should contain a Statement of Community Involvement, setting out how the community should be involved in both the continuing review of the Framework and in commenting on significant planning applications. The Statement will set the standard for good practice in engaging those with an interest in the proposed development. "... In the case of large developments, *we propose that compliance with the terms of the Statement and its requirements for engaging the community, should be a material consideration supporting a planning application.*" (emphasis added)

STEP 3: MITIGATION OF POTENTIAL ADVERSE EFFECTS

Adverse effects can be addressed either through physical mitigation measures or through the imposition of conditions that restrict the operation to the extent that the source is eliminated. The efficacy of measures intended to ameliorate any negative effects needs to be established.

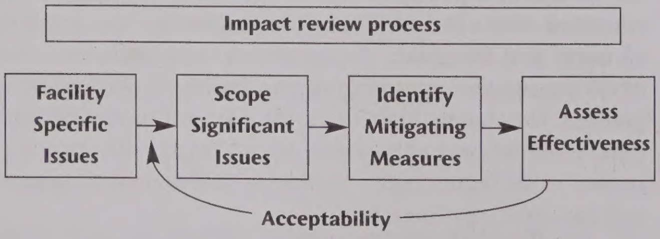
It is possible to devise a hierarchy of mitigation actions as follows:

Level of Mitigation	Example Action for Noise
Prevent the adverse effects occurring at the source	Replace reversing alarms with visual warning
Reduce the effects at source	Use silencers
Create a break in the source/receptor pathway	Construct noise bund
Reduce the effects at receptor, Repair effects, compensate or enhance	Provide double glazing Provide recreational space to compensate for loss of quiet space

There is usually a range of mitigation techniques available in relation to each level of action for each material consideration and the following issues may be considered to determine which to select:

- the conditions under which each may be most appropriately deployed;
- the expected benefits in terms of proven or demonstrable reductions in impact;
- the costs of deployment; and
- any practical issues to be overcome such as reliability.

An important consideration here is where the baseline of effectiveness of mitigation is set. Also of major significance is the extent to which any measures are reasonable and in proportion to the size of the development.



The baseline is constantly changing as operators strive to improve standards voluntarily or in response to regulatory pressure, and thresholds of acceptability change – generally downward. Accepted practice needs to be clarified so that additional burdens associated with adopting particular practices can be clearly understood. In this way a proper understanding of the relative costs and benefits of different measures can be fully appreciated. This has a close parallel to the process of identifying the Best Available Technique under the Pollution Prevention Control (PPC) regime.

CONCLUSION

The process of understanding the full implications of siting of waste management facilities is complex and requires sophistication and sensitivity to a wide range of issues. The industry and planning authorities are developing their approach, but this can tend to be on a piecemeal basis in response to concerns and may be very site specific. The ability of waste management professionals to work together, building on the positive experience and successes within the sector will help to move the debate forward, creating a platform from which progress towards more sustainable waste management can be achieved.

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